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New approaches to environmental management research at landscape and watershed scales

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Agriculture is the world's largest industry, and as a landscape-based enterprise, it has an environmental impact that extends well beyond the borders of individual fields. Water, nutrients, dust, pests, pollen—whether managed intentionally or unintentionally—leave fields and farms for points downstream and downwind, affecting organisms, water quality, and air quality at locations sometimes far distant from their points of origin. Such effects have been a defining feature of agriculture since its early days. Indeed, early atmospheric methane increases have been linked to the onset of lowland rice cultivation more than 2,000 years ago (Intergovernmental Panel on Climate Change, 2002), and carbon dioxide increases since 1750 have been caused in part by the onset of widespread land conversion to cropland following European expansion (Wilson, 1978). More recent effects, ranging from nutrient exports to groundwater to the regional delivery of windborne, micron-sized particulates, are well documented (National Research Council, 2000; Aneja et al., 2006).

Arguably, agriculture depends upon landscapes as much as agriculture affects them. Natural predators of crop pests, for example—important even where pesticides are used and crucial elsewhere—are able to colonize crops to feed on pests only where landscape features, such as woodlots and unmanaged fields, provide food, cover, and other resources that allow them to persist during those parts of the year that crops are unavailable (Landis et al., 2005). Soybean aphids in the Great Lakes region are readily controlled by Coccinellid beetles in landscapes with woodlots that provide overwintering habitat and successional fields that provide early flowering native plants for spring food. Likewise, crops depend upon landscape heterogeneity for other services: For pollinators, for climate moderation, for irrigation recharge, for windbreaks, and for mitigating the downstream effects of agriculture that would otherwise be much more severe.

These latter effects are perhaps the best studied and most purposefully managed landscape attributes. First-order streams and wetlands can transform nitrate leached from agricultural fields to more inert forms of nitrogen, such as nitrogen gas (Lowrance, 1998). Riparian buffer strips trap overland flow and its transport of phosphorus and sediment to surface waters (Richardson and Qian, 1999). Successional fields, including Conservation Reserve Program (CRP) land, provide habitat for birds and other wildlife that would otherwise be more completely displaced by agricultural conversion (Ryan, 2000; Farrand and Ryan, 2005; Johnson, 2005).

In short, agriculture is as much as ever, and perhaps more so today, a landscape enterprise. And as we move into an era in which ecosystem services from agriculture are tabulated and valued (Robertson and Swinton, 2005; Swinton et al., 2006), landscape involvement and management will become ever more important. A majority of the noncommodity services provided by agriculture involve landscape elements and landscape-level processes. Clean air, clean water, biodiversity, wildlife, and visual amenities are but a few examples.

Why, then, are agricultural landscapes understudied with respect to these processes? A variety of reasons have been cited; they range from paradigm limitations to social and scientific barriers (e.g., NRC, 2003; Robertson et al., 2004). Underlying most reasons, however, are the methodological: Too few studies have employed appropriate methodologies at the appropriate scales to provide the comprehensive knowledge needed to manage landscapes effectively for the full suite of services they can provide.

While we believe this insufficiency is regrettable, we also believe it can be overcome, and we elaborate on this belief in the pages that follow. Our intent is to provide, first, an overview of traditional field-scale approaches to environmental management and research in agriculture, highlighting both successful examples and their general limitations; second, to describe what we view as the three main challenges for incorporating landscape methodologies into the present research portfolio; and third, to define and describe what is needed in the way of new tools, approaches, and research to overcome the barriers that presently inhibit a landscape orientation and that could help address important and otherwise recalcitrant environmental problems associated with agriculture.

Traditional approaches to management and research

Successful research strategies

Traditional approaches to studying and managing environmental issues in agriculture most often are field-based, sometimes farm-based, and least often watershed-based. Field-scale approaches have proved especially useful for defining the range of management options available for solving specific field-level problems. Nutrient management and wildlife conservation are research areas that illustrate the range of successful approaches for conducting field-scale research.

The U.S. approach to nutrient management research and extension for more than 50 years has been based on statistical analyses of small-plot trials in which crop responses to several fertilizer or manure treatments are tested in a randomized and replicated experimental design. Such experiments are traditionally conducted at agricultural experiment stations, usually located on level, relatively homogeneous soils representing major soil types. Farmers and agricultural industry representatives are invited to view the results of these trials at field days and experiment station open houses. Results from those experiments are used to develop fertilizer and manure recommendations and are often published in leading research journals and experiment station bulletins. These experiments have shown the value of soil testing for fertilizer recommendations and manure testing for manure management. They also have provided valuable information about impacts of rate, timing, method of application, and formulation of various fertilizer products on crop yield and nutrient loss.

In recent years, plot-based nutrient management research has been extended to include on-farm sites. On-farm variety and tillage trials have long been used to bridge plot-level, station-based research and farmer adoption. On-farm nutrient management research likewise allows in-depth investigation across a wider variety of soils and landscape positions, questions normally limited in station-based research. While experiments at the plot scale are statistically rigorous, straightforward to design and implement, and relatively inexpensive, researchers, industry professionals, and farmers have increasingly become concerned about the difficulty of extrapolating results from small-plot experiments across the broader landscape. On-farm research provides the potential for broader extrapolation, especially with new precision-based technology. Together, plot-based experiments and on-farm research have been hugely successful in providing solutions for specific field-based nutrient management problems.

In contrast to nutrient management research, experiments using tightly controlled, small-plot designs are rarely useful in wildlife science. Although experiments at the plot scale may be appropriate for studies of vegetation or even insect response, wildlife populations typically respond to land use at much larger scales, requiring alternative experimental approaches. Moreover, what constitutes an appropriate spatial scale varies among species with different life-history characteristics. Consequently, much of the evidence for wildlife benefits of conservation practices comes from studies conducted at the practice and field scales in which management of the noncropped portion of the landscape is varied, with replication across relatively homogenously managed practice units or fields. With these designs, researchers often attempt to control for extraneous sources of variation by blocking or pairing on farm, landowner, location, landscape context, or management regime.

Simultaneous use of controls and treatments in designed experiments allows strong inference through the establishment of both necessary and sufficient causation. In contrast, the identification of appropriate no-treatment "controls" is often problematic in field- or farm-level conservation studies. The reason: It is not always clear whether a control should represent status quo conditions, agricultural production in the absence of the conservation practice, alternative implementations of the conservation practice, or baseline wildlife populations in the natural community for which the conservation practice is a surrogate (e.g., remnant native prairies versus CRP fields).

Random assignment of treatments and controls to experimental units is a hallmark of the scientific method and guards against systematic error, spatial autocorrelation, and subtle researcher bias. But unlike small-plot experiments that are often conducted on agricultural experiment stations, farm- and landscape-scale studies of conservation practices are most often carried out on private working land. As such, the researcher rarely has control over treatment assignment. For instance, most field-, farm-, and landscape-scale studies of wildlife responses to conservation practices are conducted as observational studies where previously treated experimental units (e.g., CRP fields) are randomly selected from the population of units to which the researcher wishes to make inference (e.g., all the CRP fields in a watershed, county, state, or region). Regrettably, convenience sampling too often is substituted for probabilistic sampling, which substantially impairs inferential strength.

Four lines of evidence demonstrate success of a particular wildlife conservation practice (after Ryan, 2000): (1) Occupancy of the practice by the focal wildlife species; (2) high population abundance in the conservation cover relative to alternative habitats, in particular those land uses or land covers that the conservation practice replaces (e.g., cropland); (3) reproductive success sufficient for positive population growth (i.e., λ greater than 1); and (4) positive population growth (or reduced decline) after initiation of the practice.

Despite all of its experimental limitations, extant field-, farm-, and landscape-scale wildlife research documents show overwhelming benefits of conservation practices implemented under U.S. Department of Agriculture (USDA) conservation programs, particularly the CRP (Burger, 2005; Clark and Reeder, 2005; Farrand and Ryan, 2005; Johnson, 2005; Reynolds, 2005). Moreover, those studies demonstrate that accrued benefits vary among species in relation to physiographic region; season; conservation practice and cover; time since establishment; disturbance regimes; patch size, shape, and distribution; and landscape context.

The need for new approaches

Traditional field- and farm-scale management approaches cannot in themselves provide relief from agroenvironmental problems that are expressed at regional to continental scales. As successful as many of the traditional approaches have been, they—and especially the ones that focus on increasing overall production rather than on more efficient production—suffer from limitations that restrict their effectiveness or implementation at larger scales. Mainly, this is due to factors that cannot be addressed with research designed to answer single, narrowly defined questions in the context of individual, field-based management practices.

To address problems that manifest themselves at larger scales, a different approach is needed. Experience suggests that effective solutions will share some combination of the following attributes:

- A systems orientation that balances multiple aims against known tradeoffs. It is well and good to find a specific solution to a specific problem, but ensuring that the solution does not create problems elsewhere in the landscape requires a systems orientation, especially when effects are indirect or offsite. A systems orientation also allows multiple aims to be balanced and synergies optimized.
- Geographic scalability, whereby positive effects at one scale do not disappear at larger scales. Solutions developed at small scales may not be appropriate or even applicable at larger scales. And spatial heterogeneity and

landscape complexity can affect the efficacy of many management practices and in some cases—such as wetland nitrate and phosphorus attenuation—provide solutions not available at smaller scales.

- Socioeconomic considerations that provide for solutions that can be implemented using realistic incentives at an acceptable economic and social cost. Solutions that fail to consider social acceptance or economic viability at multiple scales risk irrelevance due to lack of acceptability and, thus, implementation. Solutions obvious to the biophysical scientist or agronomist may have hidden social costs and unforeseen barriers to adoption.
- Long-term responses are included, such that climatic, social, and other factors that change on a years-to-decades time scale can be evaluated and against which management impacts can be clearly distinguished from impacts due to long-term trends in the biophysical environment, such as regional climate change. Many environmental attributes change slowly or are affected by major events at infrequent intervals. Without long observation periods that allow changes to be assessed and climatic, population-outbreak, and other infrequent events to be captured, it is difficult to develop informed, robust solutions.

Challenges to adoption of new approaches

Three main challenges confront adoption of landscape-level approaches to environmental research in agriculture: (1) The systems approach is difficult and expensive; (2) regionalization requires extensive sampling and modeling; and (3) the inclusion of socioeconomics requires a new research paradigm.

The systems challenge

Examples of systems approaches to environmental problems in agriculture are few, largely because of the geographic scale and information needs of a systems approach. Those needs make the approach time-consuming and expensive, but ultimately more effective. Two examples illustrate the challenge: Nutrient management and wetland restoration.

Nutrient management. Notable exceptions to the pattern of small-scale approaches to reduce the transport of nutrients, sediments, and greenhouse gases from agriculture are the Management Systems Evaluation Areas (MSEA) and the Conservation Effects Assessment Project (CEAP) programs. The goal of the MSEA program, funded by USDA in the 1990s, was to develop and promote agricultural management systems that reduce the impact of farming on groundwater and surface water quality. MSEA sites (plot, field, and small watershed scales) were located in Ohio, Missouri, Minnesota, Iowa, and Nebraska (Ward et al., 1994). Extensive evaluation of the water quality impacts of farming systems were conducted at those sites, where numerous best management practices (BMPs) were evaluated for their relative effect on water quality. Water quality modeling was used to predict that water quality would improve at watershed and regional scales with reduced applications of phosphorus or nitrogen fertilizers and increased adoption of soil conservation practices.

In contrast to MSEA, the CEAP program is designed explicitly to study the relationships between agricultural management practices and water quality at the watershed scale and, in particular, to evaluate the effectiveness of BMP implementation in select watersheds with a long record of water quality monitoring data (Mausbach and Dedrick, 2004). Those studies are designed to address the effectiveness of BMPs for soil erosion control and nutrient management over a wide range of soil, landscape, climatic, and land use characteristics. CEAP 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.

By considering the impact of alternative BMPs on a variety of system attributes and response variables, CEAP is more systems-oriented than most such programs. Nevertheless, it is incomplete. What constitutes a complete systems approach for nutrient management issues? A complete approach should study the system at the spatial and temporal scales for which it is possible to clearly identify relevant inputs, processes, outputs, feedback loops, nonlinearities, complexities, recovery patterns or resilience, and external controls. It should recognize that agricultural management is affected by farm, energy, financial, environmental, and health policies. It should involve the study of production characteristics; landscape features; farm manager attributes; variability in climate and hydrology; environmental impacts on soil, water, air, and health; and such issues as biodiversity, connectivity, and wildlife habitat.

Systems approaches recognize complexity. For example, a given BMP does not have the same effectiveness for improving water quality across all soil types, landscape positions, climatic regions, or management systems. A sediment BMP differs in effectiveness depending upon slope steepness, distance from a surface water body, and frequency of intense storms. A nitrogen BMP varies in effectiveness in response to such factors 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.

Systems approaches recognize nonlinearity. For example, the effectiveness of BMPs on water quality may depend upon thresholds or critical values. Reducing phosphorus fertilizer application rates, for instance, may have little impact on water quality if soil phosphorus levels are excessive, but the same reductions may have a dramatic impact if implemented on another soil with moderate soil phosphorus levels.

To complicate matters further, the effectiveness of a nitrogen BMP may depend upon both complexity and feedback loops. For example, the effect on water quality of reducing nitrogen fertilizer application rate may depend upon the amount of crop residue left behind for soil erosion control and the type of tillage practiced. Greater amounts of residue may immobilize more nitrogen, 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, depending upon the overall impact of tillage on immobilization versus infiltration.

As another example of complexity and feedback loops, consider controlled drainage, which is promoted as a method for reducing nitrate losses to surface waters. This reduction typically comes at the expense of increased emissions of nitrous oxide, a greenhouse gas. From a systems perspective, the increased greenhouse gas emissions could lead to further global warming and increased precipitation, which could potentially offset the benefits of direct reductions in nitrate emissions to surface waters from controlled drainage (Robertson, 2004).

Finally, systems approaches recognize the larger policy context in which systems outputs are interpreted. During the 1980s, farm policy was heavily influenced by the goal of increased crop production and sustainability of soil resources. Soil erosion control was discussed in the context of keeping soil losses below soil tolerance values so that there would be no long-term reductions in crop productivity. Currently, soil erosion control is being discussed in terms of policies whose goal is to maintain crop productivity while protecting environmental quality at a level assessed by aquatic goals [e.g., total maximum daily loads (TMDLs), concentrations of total suspended solids (TSS), substrate embededness, and vigor of submerged aquatic vegetation]. In this context, it makes little sense to find management solutions that reduce erosion to soil tolerance values because more dramatic reductions are generally needed. The larger reductions needed are producing a wider range of management options for soil erosion control than conservation tillage, including riparian buffer strips, cover crops, alternative cropping systems, and land retirement.

Wetland restoration. One of the most important changes in agricultural landscapes that has taken place as a result of conservation programs in the last decade is the restoration of wetlands. This has been accomplished through various USDA and other programs. Most wetlands restored through these programs are former wetlands in which hydrology had been altered to make crop production feasible. Although not generally applied to restored agricultural wetlands, functional assessment techniques are available to determine when wetland functions have been restored (Brinson et al., 1995; Brinson and Rheinhardt, 1996). Functional assessment of wetlands has been driven by section 404(d) of the Clean Water Act, which has limited the draining and filling of wetlands. Functional assessment techniques have been developed to guide wetland mitigation. Those techniques are also applicable to voluntary, incentive-based wetland restoration.

Functional assessment techniques depend upon comparison of wetland attributes to those of a reference wetland (Brinson and Rheinhardt, 1996; Findlay et al., 2002). Lists of wetland functions (e.g., Table 1) have been developed for a variety of wetland types. Those assessments have been driven by both wetland protection and mitigation. One major impediment to understanding the functional restoration of wetlands is that functional assessment depends in large part upon the comparison of restored wetlands to reference wetlands. In many agricultural landscapes, there are no reference wetlands available because of changes in hydrology and land surfaces (geomorphology) associated with the original conversion of wetlands to arable farmland. Without reference wetlands, functional assessments must depend upon a few key factors rather than a suite of factors.

One of these key factors is wetland hydrology, which illustrates the danger of using one factor alone to assess wetland function. Examples abound of hydrologically restored but disturbed wetlands that have been overtaken by exotic invasive plants, such as Brazilian pepper, purple loosestrife, and reed canary grass; as a result, those wetlands have few functional attributes of the original wetlands they were intended to restore (Ferriter, 1997; Zedler and Kercher, 2004).

Biogeochemical budgets can also be used to assess functionality. Such budgets provide an integrated measure of system performance and thus can be used to evaluate the system-level response to restoration efforts. But long-term monitoring and large amounts of resources are necessary to develop hydrologic budgets, and the expense is even greater when nutrient and sediment budgets are pursued. Nevertheless, wetland restorations and creations have been evaluated using input-output budget techniques on a scale ranging from a few hectares or acres to many square kilometers or miles (e.g., Kovacic et al., 2000; Vellidis et al., 2003).

The purposes of wetland restoration efforts funded under USDA's Conservation Reserve Enhancement Program (CREP) may differ, even in states with similar problems. In the U.S. Corn Belt, for example, the Iowa CREP specifically focuses on reduction of nitrate from tile drains. Wetland siting is based on the primary criterion of controlling nitrate movement from drainage systems (Figure 1). The effectiveness of those wetlands are evaluated by following long-term changes in watershed nitrate transport. The greatest potential benefits for nitrate mass reduction will be in extensively row-cropped and tile-drained areas of the Corn Belt where the nitrate loads are highest (Crumpton, 2000, 2005). But removal rates are not only driven by nitrate loading; hydrologic loading and water residence times are important as well (Kovacic et al., 2000).

In contrast, the Illinois CREP is focused on a wider array of goals, including sediment load reduction; nitrogen and phosphorus load reductions; increased populations of waterfowl, shorebirds, and threatened and endangered species; and increased native fish and mussel stocks. Establishment of wetlands for a wider range of purposes that are more consistent with general stream restoration goals is likely to increase the potential of restoration sites. But evaluation of programs with more complex criteria is more difficult than the evaluation of those with more simple goals.

The regionalization challenge

The second challenge facing incorporation of new approaches is that of regionalization, which requires extensive sampling and extrapolation that, for a variety of reasons, can be difficult. The management of nitrate, sediment, and wildlife illustrate these issues.

Management of nitrate on a landscape scale. High concentrations of nitrate in tile-drain systems lead to flow-weighted concentrations of nitrate above 10 milligrams of nitrate-nitrogen per liter in many streams draining the agriculturally important region of the Corn Belt. Artificial subsurface drainage causes water to move through the root zone much faster than in undrained land. Because of this increased rate of movement, nitrate is leached to the drain system and then moves out to receiving waters. This phenomenon occurs in most environments where agriculture is conducted on drained soils: The Southeast (Lowrance et al., 1984), the Lake Erie Basin (Calhoun et al., 2002), Ohio (Tan et al., 2002), and Quebec (Elmi et al., 2004). Although most nitrate in agricultural landscapes is derived from fertilizers, it

Table 1. Functional characteristics of riverine wetland classes used to assess wetland restoration success.

Class	Functional attribute
Hydrologic	Dynamic surface water storage
	Long-term surface water storage
	Energy dissipation
	Subsurface storage of water
	Moderation of groundwater flow or discharge
Biogeochemical	Nutrient cycling
	Removal of imported elements and compounds
	Retention of particulates
	Organic carbon export
Plant habitat	Characteristic plant communities
	Characteristic detrital biomass
Animal habitat	Spatial structure of habitat
	Interspersion and connectivity
	Distribution and abundance of invertebrates
	Distribution and abundance of vertebrates

Source: Brinson et al., 1995.

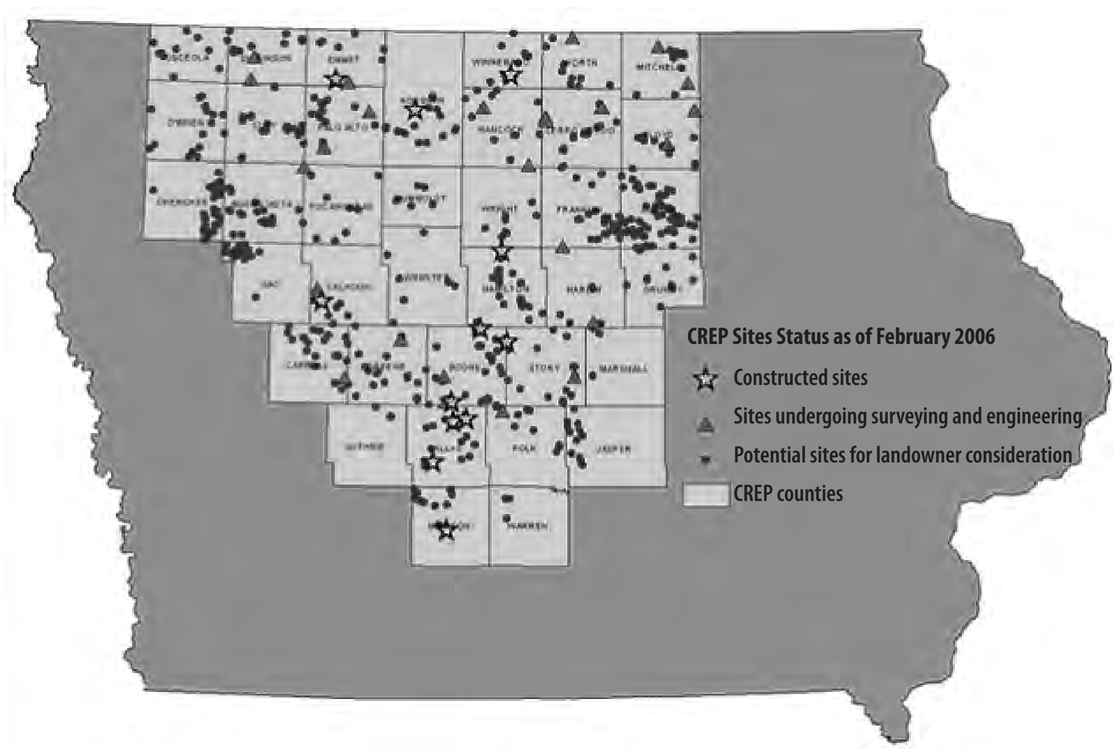

Figure 1. Sites for nitrate removal wetlands identified for the Iowa Conservation Reserve Enhancement Program (CREP). Source: Iowa Department of Agriculture and Land Stewardship, Water Resource Bureau (http://www.agriculture.state.ia.us/CREP.htm).

can be derived from any nitrogen source, including soil organic matter, manure, and atmospheric deposition.

The effects of excess nitrate on environmental health are generally manifested through enhanced algal growth in downstream water bodies, eventually leading to lowered levels of dissolved oxygen and other signs of eutrophication, depending upon nitrogen:phosphorus ratios in freshwater or nitrogen:phosphorus:silica ratios in coastal waters.

Management of nitrate is tightly tied to management of the nitrogen cycle, which has myriad biological and chemical reactions that affect nitrogen oxidation state and transportability. Because different landscape components can be structured and managed to encourage these reactions, there are many landscape-scale management options for nitrogen in general and nitrate in particular. Measurement of the effects of these landscape management options can range from sophisticated techniques tied to gaseous evolution, to techniques to understand the movement and retention time of water, to techniques to understand the relative areas of cropland versus wetland. In tile-drained landscapes, measurement of changes in nitrate transport in response to landscape and agricultural management is relatively straightforward.

Although measuring nitrate levels in water is straightforward, interpreting the effects of landscape management or adoption of nitrogen-conserving practices on a landscape scale is difficult. Numerous studies, for instance, have shown that in the Corn Belt a switch from fall application of anhydrous ammonia to spring application of other nitrogen sources (e.g., urea ammonium nitrate) will lead to 15 to 20 percent reductions in nitrate concentrations (and loads) in tile drainage water (Randall and Sawyer, 2005). But documenting the change in nitrate concentration and load on a landscape or watershed scale for this relatively simple nitrogen conservation technique has been difficult, largely because of the inherent difficulty of paired watershed studies and comparing watersheds over time.

Jaynes et al. (2004), for example, found that in a 400-hectare (1,000 acres) treated watershed, where most farmers had switched to spring nitrogen application (late spring nitrate test), average annual flow-weighted concentrations were lower than in two control watersheds, where farmers did not switch from fall anhydrous applications. The study was carried out for five pre-treatment and four post-treatment years (1992-2000), and differences between the treated watershed and the two controls were only evident in the last two years of treatment (1999 and 2000) (Table 2). Study results were complicated by the finding that one of the control watersheds consistently had the lowest concentrations in the pre-treatment period and by the finding that nitrate concentrations in the treated watershed only declined about 1 milligram of nitrate-nitrogen per liter during the four post-treatment years compared to the pre-treatment periods. Nevertheless, Jaynes et al. (2004) concluded that the late-spring-nitrogen-test treatment resulted in a 30 percent or greater reduction in nitrate-nitrogen in the drainage water in the last two years of the treatment period (Figure 2), with no reduction in yields. Although results showed that the late-spring-nitrogen-test-treated subbasin had lower nitrate concentrations, the measurements showed the difficulty of demonstrating the efficacy of nitrogen-conserving practices on a watershed scale. In two years of the treatment period (1997 and 1999) actual nitrogen reductions on the treated watershed were small or nonexistent. In two years (1998 and 2000), the fertilizer rates were greatly reduced [by 46 and 73 kilograms (41 and 65 pounds) of nitrogen per hectare per year]. Given the difficulty and expense of these types of watershed-scale studies, it is not surprising that there are few other published studies of this sort.

Phosphorus and sediment loads at landscape scales. Efforts to assess BMPs to reduce phosphorus and sediment loads from watersheds illustrate different approaches to watershed assessment. Traditional approaches typically involve water quality monitoring at the mouth of the watershed, before and after implementation of BMPs within the watershed. This approach is limited in its ability to study cause-and-effect relationships because water quality monitoring data reflect cumulative impacts of large areas with different landscape features, management practices, vulnerabilities to pollution export, and climatic patterns. Furthermore, long-term water quality data sets (greater than 10 years) are typically needed to identify trends in water quality. An example of the latter is

Table 2. Flow-weighted average annual nitrate-nitrogen (NO3-N) concentrations in water draining Iowa watershed sub-basins in which fields were nitrogen-fertilized as usual (control 1 and 2) or nitrogen-fertilized on the basis of late spring nitrate tests (LSNT) beginning in 1997 (TR1).

Subbasin	Control 1	Control 2	LSNT
1992 (mg NO_3-N L^{-1})	9.9	13.7	12.5
1993 (mg NO_3-N L^{-1})	8.2	9.7	9.2
1994 (mg NO_3-N L^{-1})	9.2	10.2	8.9
1995 (mg NO_3-N L^{-1})	13.1	16.7	16.0
1996 (mg NO_3-N L^{-1})	14.0	15.4	15.6
1997 (mg NO_3-N L^{-1})	8.4	13.1	10.8
1998 (mg NO_3-N L^{-1})	11.1	14.0	10.2
1999 (mg NO_3-N L^{-1})	15.8	16.5	11.7
2000 (mg NO_3-N L^{-1})	16.5	15.1	11.0

Source: Jaynes et al., 2004.

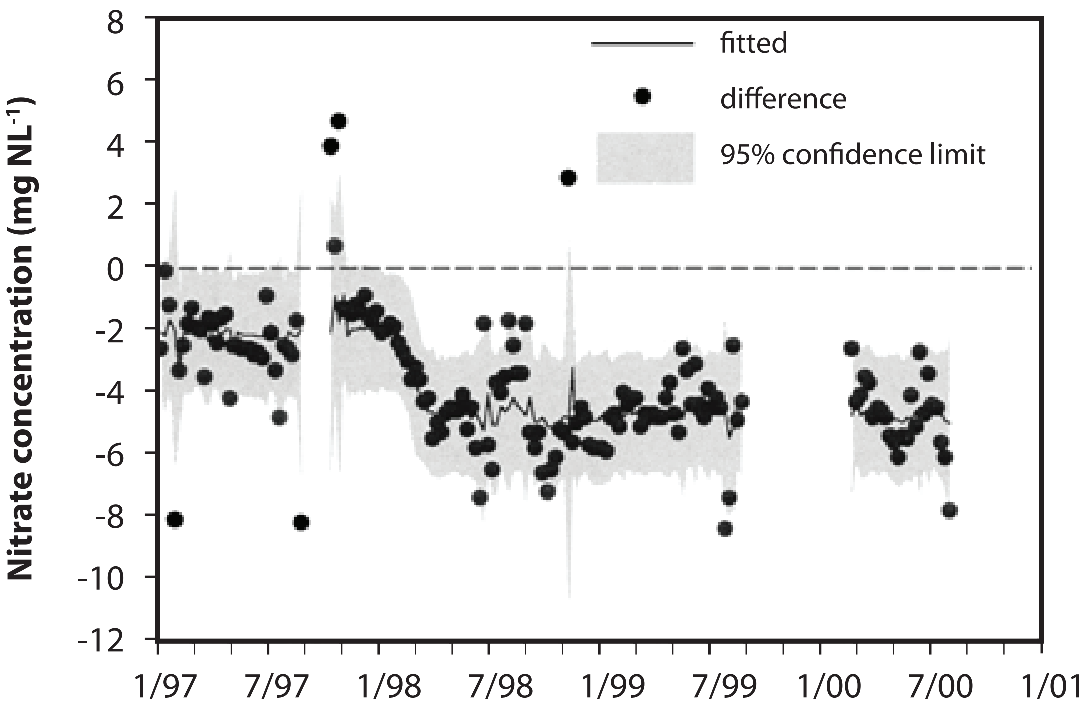

Figure 2. Change in flow-weighted drainage nitrate levels from a watershed subbasin in which fields were nitrogen-fertilized based on late season nitrate test (LSNT) relative to levels from a business-as-usual subbasin. Source: Jaynes et al., 2004.

a study conducted by Richards and Baker (2002) on four watersheds in Ohio. They studied water quality data from 1975 to 1995 using analysis of covariance, with time and seasonality as covariates. Significant reductions were observed in total phosphorus and total suspended solids, but not nitrate-nitrogen.

A second approach to watershed regionalization is water quality monitoring upstream and downstream of the area where BMPs are 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, however, if the upstream monitoring station collects water from a very large area because it will be difficult to detect small changes in water quality due to implementation of BMPs downstream.

A third approach is multiyear monitoring of multiple watersheds where BMPs have been implemented. A challenge, however, is normal variability in river flow. It is difficult to separate the influences of flow variation due to climatic variability from the effects of BMPs. Davie and Lant (1994), for example, studied the impact of CRP implementation on sediment loads in two Illinois watersheds. They found that CRP enrollments on 15 and 27 percent of cropland reduced estimated soil erosion rates by 24 and 37 percent, respectively, but sediment loads at the mouths of the watersheds declined by less than 1 percent. They attributed those small overall impacts to poor targeting of CRP to land in close proximity to streams and to a time delay in sediment transport from the field edge to the mouth of the watershed.

The most rigorous approach to watershed-based regionalization involves paired watershed comparisons, as in the Jaynes et al. (2004) nitrate example above. 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 in northern Missouri. After a seven-year calibration period, grass and agroforestry strips were initiated and found to reduce total phosphorus by 8 and 17 percent, respectively, during the first three years. The contour strip and agroforestry treatments reduced runoff by 10 and 1 percent, respectively, during the treatment period. The contour-strip treatment reduced soil erosion by 19 percent in 1999, while soil erosion in the agroforestry treatment exceeded the predicted loss.

Birr and Mulla (2005) implemented conservation tillage on 70 percent of the moldboard-plowed acreage for three years in a 1,100-hectare (2,750-acre) watershed in southern Minnesota. No changes in tillage were made in an adjacent watershed. Although these changes resulted in a 40 percent reduction in soil erosion for the treated fields and an estimated 20 percent 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. This outcome, according to the researchers, was 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.

Wildlife management at larger scales. The basis of our understanding of wildlife benefits of conservation practices has been derived primarily from studies conducted at the patch, field, and farm scales, and generalizations to larger spatial scales are difficult because of the substantial variability and often contradictory nature of observed responses. Apparent contradictions are frequently attributable to variation in landscape context. Both landscape composition and structure can influence ecological processes, such as dispersal, predation, habitat selection, and population performance, and, hence, observed responses.

In central North Dakota, for example, daily survival rates of upland-nesting ducks exhibited a curvilinear relationship with patch size, with lowest survival occurring in intermediate-sized patches (Horn et al. 2005). Moreover, landscape composition (percentage of landscape in grassland habitats) altered the functional relationship between daily nest survival and distance to nearest field edge. Those differences in predation patterns may be attributable to effects of landscape composition on predator habitat selection (Phillips et al., 2003), space use (Phillips et al., 2004), and foraging efficiency.

Thompson et al. (2002) hypothesized that effects of habitat fragmentation operate within a spatial hierarchy in which effects at larger scales (regional and landscape) impose constraints on local scale (patch and edge) effects. Consistent with this hypothesis, Stephens et al. (2003) reported that studies where habitat fragmentation was measured at landscape scales were more likely to detect effects on avian nest success than studies measuring fragmentation at local spatial scales. Similarly, Chalfoun et al. (2002) reported that studies conducted at larger spatial scales were more likely to detect functional and numerical responses of nest predators to fragmentation. Horn et al. (2005) concluded that design of conservation cover configurations that meet conservation goals required an understanding of the patterns of nest success and the predation processes that produced observed patterns. Thus, studies designed to evaluate or monitor wildlife responses to conservation management systems need to be explicitly hierarchical in design, executed at larger spatial scales than previously considered, and conducted across a range of landscape compositions and configurations.

A recent CEAP review (Soil and Water Conservation Society, 2006) acknowledged that estimates of environmental effects of conservation programs should be compared to established environmental goals and linked to the ecological context in which the estimated effects occur. Moreover, the panel conducting the review strongly emphasized that simulations and extrapolations must not substitute for on-the-ground monitoring and inventory systems designed to determine if anticipated conservation and environmental benefits are being achieved. The panel specifically endorsed a hierarchical assessment model based on project-level monitoring within the context of regional assessments, and further suggested that a national assessment produced by aggregating valid regional assessments (context-sensitive measures) is more credible and meaningful than generalizing from national to regional scales.

The socioeconomic challenge

Agricultural activities are integrally linked with the nation's ecosystem services and the environmental health of its natural resource base. In previous sections we argued that the complex interactions between land use, water, soil, and air mandate that scientists adopt a systems approach at the landscape or watershed scale to understand the relationships between agricultural practices and the resulting ecosystem services. To appreciate the scientific need for a paradigm that integrates social and economic influences with the biophysical, it is necessary to emphasize one more feature of the system: The critical role of human actions.

The central United States provides a textbook example of fundamental antrhopogenic change in the hydrologic system and environmental resources in the agricultural landscape. The Mississippi River now carries 15 times more nitrate than any other U.S. river and twice as much phosphorous compared to loads prior to European settlement. More than 95 percent of the original prairie, savanna, and woodland in the Mississippi River watershed has been converted to agricultural use, and farmers actively manage about 75 percent of the land area within the region.

Human behavior has caused these changes. Thus, our attempt to understand the system will be fundamentally incomplete if it fails to incorporate human decision-making factors, such as economic influences, human preferences, and social mores. In short, science that wishes to inform and support policymakers and land use managers must also focus on human behavior and the links from that behavior to land use, ecosystem services, and back again. For example, biophysical scientists may identify the ideal location for reintroduction of a native species, but if economic factors lead to incompatible land uses on neighboring tracts, the reintroduction may be doomed before it begins. Likewise, limnologists may determine that dredging a lake is a cost-effective way to improve lake clarity, but if economic incentives direct farming practices to continue unchanged in the watershed, the lake may soon revert to its murky status.

Perhaps an even more compelling case for a paradigm that explicitly incorporates human behavior comes from the information needs dictated by the current policy environment of agricultural and environmental decisions. Simple observation suggests that the United States has adopted a fundamentally different approach to controlling environmental alteration associated with agricultural activities than in the case of other major industries. Specifically, air and water emissions from industrial and transportation sources are typically controlled by regulations that permit certain levels of emissions, economic fines when emission levels are exceeded, or mandatory caps with trading programs. In contrast, environmental effects associated with farming activities have been addressed primarily through voluntary actions and financial incentives. Major cost-share programs, such as the Environmental Quality Incentives Program (EQIP), the CRP, and the Conservation Security Program (CSP), among others, have pumped billions of dollars into voluntary conservation efforts. With some exceptions [e.g., confined animal feeding operation (CAFO) regulations], there is little meaningful discussion about shifting to a large-scale regulatory paradigm for agricultural conservation policy.

In such a context, federal and state governments, environmental organizations, and all those interested in ecosystem services from agriculture will need information on how best to spend their limited conservation dollars to induce changes on the landscape that most effectively meet their goals. For example, consider a nongovernmental organization or governmental entity interested in purchasing land within a watershed to locate a wetland to improve downstream water quality. A wetland expert may be able to identify ideal hydrologic conditions and spatial features to rank order the sites that would be most effective for this goal. Subsequent action might include purchasing and building a wetland on the first site from the list, then, if funding permits, doing the same for remaining sites. Following this approach, will the nongovernmental organization have done the most that it can for water quality? Not necessarily. Depending upon the cost for each of the sites, including the purchase price and restoration costs, greater water quality improvement might well be had by choosing two sites ranked lower on the list that in combination cost the same as the top site, but together produce greater benefits. The best solution becomes even more complicated if multiple land uses for each site are possible (e.g., differing perennial covers or working land in conjunction with various BMPs) and multiple environmental benefits are sought (water quality and wildlife habitat).

There are numerous other examples of policy questions that only an integrated biophysical-social science paradigm can fully address: What are the tradeoffs between focusing conservation budgets on land retirement (such as the CRP) relative to increasing conservation practices on land still in production? How much more can water quality be improved by targeting conservation dollars on the most beneficial conservation practices and locations within a watershed relative to more equally distributed conservation dollars? How can conservation programs be designed to most effectively improve environmental quality (bidding systems versus uniform payments)? How do different environmental targets (water quality, biodiversity, carbon sequestration) affect the optimal choice of land to enroll in a conservation program? What is the magnitude of the tradeoff between environmental improvement and farm profitability? How effective would a watershed trading program be in meeting water quality goals while simultaneously keeping costs low?

These are just some of the policy-relevant questions that an integrated paradigm can address. Questions of this type can only be considered within an integrated paradigm. Neither a pure social science nor a pure biophysical science approach will do.

Water quality and carbon sequestration. The first link in developing interdisciplinary paradigms that include socioeconomic factors is to develop models that predict adoption of conservation practices and land use change in response to key drivers, such as land characteristics, agricultural productivity, landowner characteristics, and costs and returns to alternative land uses. Indeed, economists and other social scientists have developed a large literature addressing the factors that induce or discourage landowners from adopting various conservation practices and/or land use decisions. For example, Sunding and Zilberman (2000) reviewed much of the economic literature on farmers' adoption of conservation practices and other new agricultural technologies. A complementary literature is quickly developing that considers the key drivers of land use decisions using spatially explicit models in conjunction with geographic information system (GIS) data to consider patterns of urban development, siting of noxious facilities, and spotty rural development. While many of these studies consider the socioeconomic questions relevant to adoption of conservation practices and/or land use change, many do not consider the biophysical consequences of

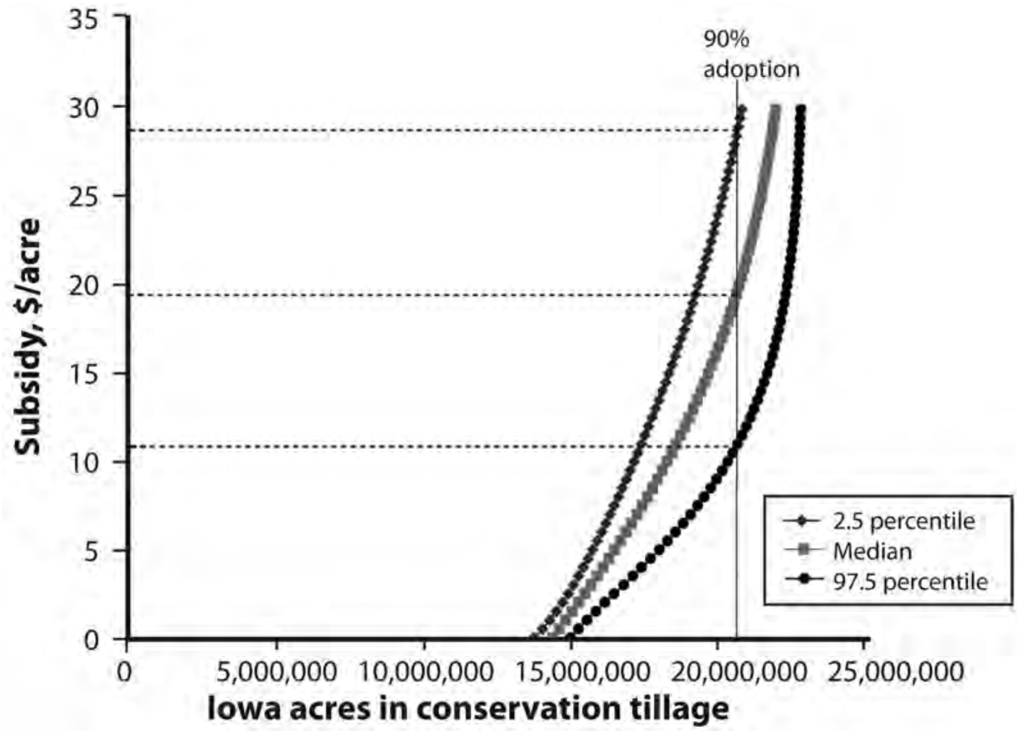

Figure 3. Estimated supply of additional conservational tillage acreage as the per acre subsidy payment increases. Source: Kurkalova et al., 2006.

changes. They fall strictly within the social science paradigm.

Kurkalova et al. (2006), for example, used observed behavior by farmers to quantify the adoption of conservation tillage practices by those farmers. Pairing these data with information on net returns to farming with conservation and conventional tillage, soil and land characteristics, weather patterns, and farmer characteristics, the authors produced a model that predicts farmers' adoption of conservation tillage as a function of its costs and suitability for their location. Using this model, the authors estimated the increased adoption that could be induced by a green payments program that paid a fixed fee for additional adoption of reduced tillage methods. The resulting "supply" schedule of conservation tillage (Figure 3) provides the link between human behavior and land use. Models of this sort can predict how acreage in a conservation practice might change with price and subsidy payment changes, but they cannot predict the consequences of these changes on environmental indicators without an explicit linkage to a biophysical model.

A number of studies incorporate simple biophysical impacts, generally measured in terms of "edge-of-field" environmental gains (e.g., Russell and Shogren, 1993). A few studies have taken the next important step by integrating economic modeling with watershed-based models (Braden et al., 1989; Carpentier et al., 1998; Qiu and Prato, 1999; Ribaudo et al., 2001; Johansson et al., 2004; Petrolia et al., 2005). To demonstrate the development and implementation of a study that incorporates economic behavior in conjunction with a biophysical model, we draw upon two recent studies that focus on conservation policy in the context of water quality and carbon sequestration, two ecosystem services that are likely to drive much of the conservation planning in agriculture in the coming decades. In each of these studies, the authors developed a behavioral model that predicts how landowners or farmers will change their land use (cropping patterns, conservation practices, and land retirement) as a result of changes in economic conditions (such as conservation payments or prices). The behavioral model is directly linked to a biophysical model that predicts how changes in land use will affect the ecosystem service.

Feng et al. (2007) considered the tradeoffs implicit in the design of a conservation program that targets carbon sequestration versus one that targets a water quality indicator, such as soil erosion. They studied only a single land use change: The choice of landowners to enroll in a land retirement program, such as the CRP, but cover a relatively large spatial area (the Upper Mississippi River Basin, which covers portions of seven states). Using the National Resources Inventory (NRI) as their unit of analysis, they linked the Environmental Policy Integrated Climate (EPIC) model to a simple economic model of landowner decision-making about the enrollment decision. There were more than 40,000 cropland "points" in their data set for which they simulated landowner decisions under a variety of hypothetical payment levels.

Findings (Table 3) suggest that policy designed to both reduce soil erosion from the enrollment of land into a CRP-type program and increase carbon sequestration in the soil will face a substantive tradeoff in choosing which of the two environmental indicators to target. For example, assuming a program budget of $500 million, a policy designed to achieve the highest level of carbon sequestration can sequester 2.9 metric tons (3.2 million tons) of carbon by enrolling about 1.5 million hectares (3.75 million acres) of land (Table 3), assuming that policy allows the authority to enroll the land that generates the highest carbon sequestration benefits per dollar spent (similar to the bidding method in the current CRP). This enrollment choice also results in sizable soil erosion control benefits. But if the program were specifically designed to target soil erosion and land was enrolled so that the highest erosion control benefit was achieved, different land would be enrolled at the same budget level. In this case, about 25 percent as much carbon would be sequestered as in the first program, but more than five times as much soil erosion would be eliminated (Table 3). From this sort of analysis it becomes possible to evaluate tradeoffs among different environmental policy decisions and optimize for desired national outcomes.

In a related paper, Feng et al. (2006) used a similar modeling framework to address a different policy question: How does a program that pays farmers to adopt conservation practices on working land (land that stays in agricultural production) affect the cost-effectiveness of a land retirement program? Landowners face an array of choices with respect to conservation programs, including those that provide payments for complete retirement of land (CRP) and those that cost-share or provide payments to adopt conservation practices on working land (CSP). When faced with mutually exclusive alternatives, farmers must make a decision, and their choice whether to enroll in the CRP, the CSP, or not participate in either program will determine the success and cost-effectiveness of each program. Again, an integrated economic and biophysical modeling paradigm allows analysts to consider these questions and provide information that can be used to improve the design of those programs.

An important shortcoming of these examples is that the biophysical model employed, and, therefore, the resulting solutions to the conservation policy choices, measures environmental effects at the edge of each field. It does not capture the interacting effects of land use and offsite water quality. While still a nascent literature, there are also some careful studies that consider the full link from proposed policy enactment to behavior change to in-stream environmental consequences (Khanna et al., 2003; Beaulieu et al., 1998; Kling et al., 2006).

The importance of disproportionalities. There is increasing recognition that nonpoint-source agricultural pollution, especially from sediment and phosphorus, arises from small fractions of the landscape (Gburek et al., 2002). Disproportionalities arise from several factors, including climatic and hydrologic patterns, topographic features, and management factors.

Birr and Mulla (2005) documented such disproportionality in two adjacent Minnesota watersheds in which they surveyed 24 farmers managing 220 fields. Results showed large disproportionalities in applied phosphorus from fertilizer and manure, ranging from 9 to 545 kilograms (8 to 487 pounds) of phosphate per hectare across corn fields, with distributions highly skewed to higher application rates on small fractions of the land base (Figure 4). A phosphorus index approach to evaluate risks of phosphorus loss to surface waters showed that 18 percent of the

Table 3. Distribution of predicted multiple environmental benefits from a uniform subsidy of $500 million for land retirement targeted toward specific benefits.

Target benefit	Benefit distribution			
	Carbon sequestered	Erosion avoided	N runoff avoided	N leaching avoided
Carbon (106 metric tons)	3.2	0.8	0.6	1.0
Erosion (106 metric tons)	7.4	40.5	14.1	9.7
N runoff (103 metric tons)	2.8	5.1	11.7	2.8
N leaching (103 metric tons)	10.0	6.4	5.6	30.6

Source: Feng et al., 2007.

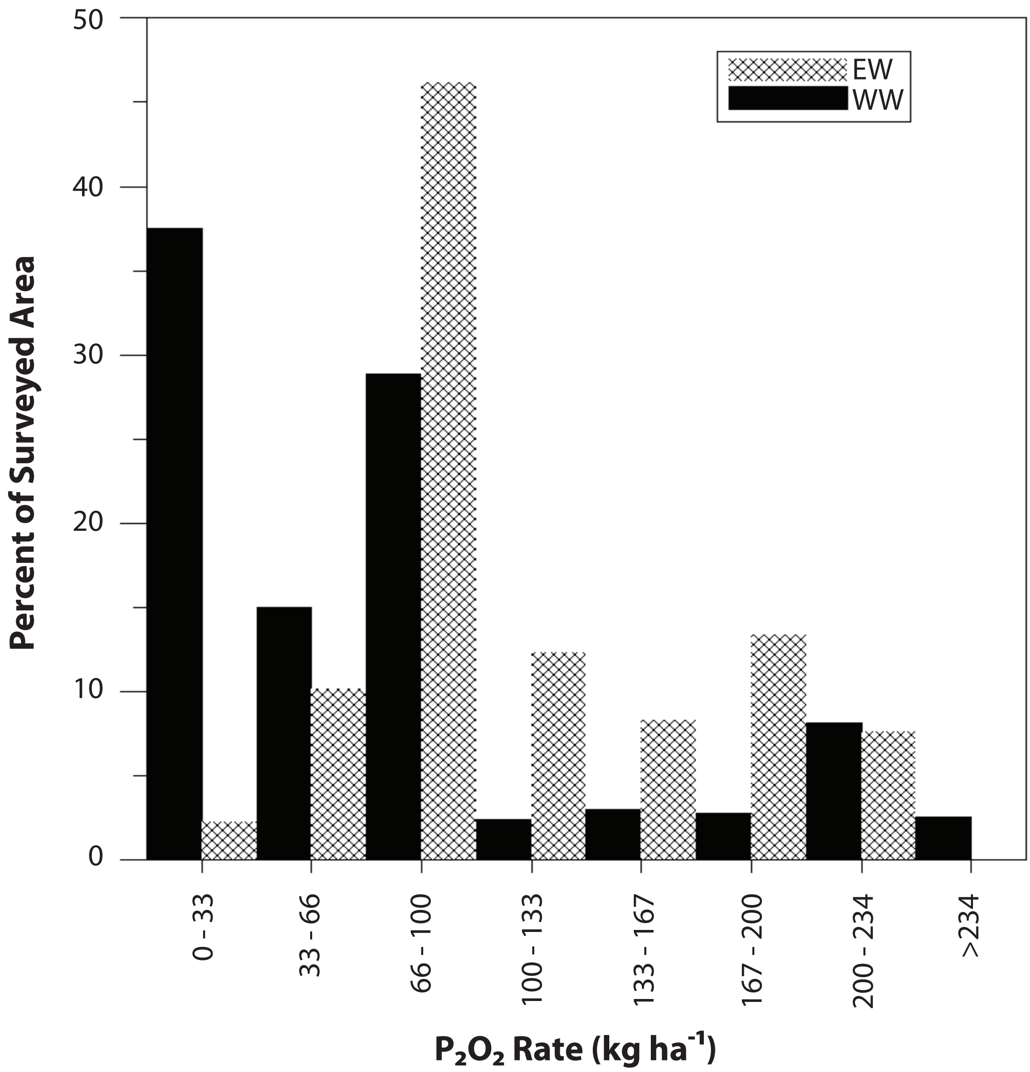

Figure 4. Disproportional distribution of phosphorus application rates on corn preceded by soybeans in a Minnesota watershed. Source: Birr and Mulla, 2005.

fields generated a disproportionate 41 percent of phosphorus losses. Those types of disproportionalities are rarely considered in watershed remediation plans because comprehensive farm surveys are rarely conducted.

Methodological needs

Underlying the difficulties of applying new approaches to researching environmental problems in agriculture are methodological limitations. Four major needs exist for adopting a systems approach that is geographically scaleable and includes an appropriate biophysical-social science balance: geospatial data, sensor networks, modeling, and long-term experiments and assessments.

Geospatial data coverage

The ability to understand and predict environmental phenomena at landscape and regional scales requires information about the landscape and region under study. Those data must be appropriate to the questions addressed and sufficiently resolved to allow for necessary extrapolation.

Data needs are to some extent discipline-specific, but a systems approach demands that data be available for a variety of questions: Land use, terrain, climatic, soil, and crop management attributes, among others. Carbon sequestration efforts can use this information to construct spatially explicit erosivity, aggregate stability, and decomposition indices. Wildlife conservation efforts can use this information to construct spatially explicit percent-cover habitat edge indices and fragmentation corridors. Twedt and Wison (2005), for example, developed a spatially explicit GIS model to guide landscape-level reforestation in the Lower Mississippi Alluvial Valley, based on habitat objectives for forest interior songbirds. Proximity to extant forest tracts, connectivity, and forest core area were important predictors in this model, such that the model's accuracy depended directly upon the availability of accurate land cover data.

Hydrologic transport models illustrate the level of detail required. As noted earlier, small areas of the landscape can contribute disproportionately large amounts of sediment and phosphorus to surface waters. These small areas are often referred to as critical areas. Critical areas are characterized as hydrologically active and connected to surface waters, and they often have inappropriate management practices on steeper landscapes. This suggests that critical areas can be identified through a combination of terrain analysis, soil hydrologic pathway analysis, and farm management surveys.

The most common terrain indices for identifying critical areas are slope steepness, slope curvature, the compound terrain index (Moore et al., 1991; Gallant and Wilson, 2000), and the stream power index (Moore et al., 1993). Each terrain index is indirectly related to landscape hydrology. Slope steepness affects runoff and surface flow velocity. Slope curvature affects convergence or divergence of runoff. The compound terrain index indicates the potential for accumulation of soil water and locations where ponding may occur. The stream power index indicates the erosive power of surface runoff.

The accuracy of terrain indices is dependent upon 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 elevation; (3) the algorithms used to calculate terrain attributes; and (4) the topography of the landscape (Theobald, 1989; Chang and Tsai, 1991; Florinksy, 1998). In general, higher resolution elevation data, such as the data available from laser imaging detection and ranging (LIDAR) systems, allow for improved identification of critical areas compared with the lower resolution elevation data typically provided by the U.S. Geological Survey.

To date there has been little research on combining terrain index modeling with soil hydrologic modeling. Surface runoff is a function of both terrain and soil attributes, but only terrain attributes are considered in indices, such as the compound terrain index and stream power index. More research is also needed on methods for combining farm management practices and terrain indices to obtain better indices of landscape hydrology.

Sensor networks

Huge amounts of data are gathered on agricultural land each year, primarily by producers. Those data are primarily yield and soil tests. Few of these data are stored in a way that can be accessed to examine spatial or temporal trends.

On more highly engineered landscapes, it will be more feasible to design and implement sensor networks to monitor and facilitate management of an environmental resource.

Investment in a sensor network will be most warranted where certain criteria are met. First, there must be an achievable management objective that requires sensor data. Second, the sensors need to be simple and sufficiently robust for routine remote field deployment. Third, assuming that the management issue is at a regional or basin scale, the sensors need to provide data that can be used in a model to provide an integrated assessment to guide real-time or near-real-time management options.

There are few places in the United States where this sort of monitoring and management occur. One example is located in the San Joaquin River drainage in California (www.sjd.water.ca.gov/waterquality/realtime/). This effort is motivated by concern for anadramous fish populations and movement of selenium from agricultural land and wetlands into coastal waters. The main objective of the project is to facilitate control and timing of wetland and agricultural drainage to coincide with periods when dilution of flow is sufficient to meet salinity objectives. Salinity is measured at numerous points in the watershed by electrical conductivity probes. Those probes are relatively simple and robust, and they provide real-time data on downstream salinity and on salinity from agricultural land and wetlands in the San Joaquin River drainage. By increasing the frequency of meeting downstream salinity objectives, the project seeks to reduce the number and/or magnitude of releases of high quality water made specifically to meet downstream salinity. The high quality water saved can be used later to increase streamflow in the basin during critical periods for anadromous fish restoration efforts.

Modeling

"All models are wrong, some are useful." This well-known adage, attributed to physicist George Box, may be more relevant in the realm of integrated socioeconomic and biophysical models than most. To provide answers to questions related to agroenvironmental policy, we must develop integrated landscape-scale or watershed models that capture complex interactions among land use, conservation practices, land characteristics, and ambient water quality. The models must also couple those factors with land use choice models that incorporate economic and social drivers of choices. A number of studies have contributed to progress in this arena (e.g., Khanna et al., 2003; Beaulieu et al., 1998), but significant challenges remain. The overall need is for models that adequately integrate the biophysical and socioeconomic dimensions of important environmental issues. Five specific needs include the following:

1. Scale compatibility.
2. Adequate data coverage for model estimation and calibration.
3. Complete model coverage of policy-relevant choices.
4. Treatment of uncertainty within and across models.
5. Development and implementation of optimization algorithms using models.

Scale compatibility. Ideally, the appropriate choice for the spatial and temporal scale of a study depends upon the questions addressed. For example, an analysis designed to identify watersheds that would be best targeted for conservation funding, such as the CSP, might reasonably be based on aggregate units. A reasonable basis for selecting a watershed for inclusion in the program might be that average environmental benefits from conservation practice adoption are higher than in other watersheds and average costs are similar or lower than in other watersheds. Once targeted for funds to identify the locations that would yield the most environmental gain for the funds allocated, however, requires a much more spatially detailed modeling paradigm—one based on data from fields or farms.

Often, the choice of scale at which modeling takes place is limited by the availability of appropriate data and the capacity of computing technology. Riffell and Burger (2006), for example, used NRI data to evaluate CRP effects on northern bobwhite and grassland bird populations nationally. NRI data provide spatially explicit, point-level information on land use, but CRP practice data is low resolution, broadly documented as grass-legume or trees, with no information on year of enrollment, specific CRP practice, or patch size or shape. In contrast, USDA's Common Land Unit (CLU) polygon data, which contain information about enrollment year, conservation practice, and field boundaries, allow detection of more specific responses to CRP vegetation type, age of planting, and landscape configuration when coupled with National Land Cover Data. CLU data were available for only a two-state subset of the bobwhite range, however (Riffell and Burger 2006).

Researchers must also trade off the increased detail that comes from working at fine scales (plots or fields, hours or days) with the ability to handle very large data sets (e.g., the number of fields in the Corn Belt or the number of hours in a decade). Even in these cases however, a fairly aggregate scale may still generate conclusions that are valuable to the decision-maker, although it will be important to recognize the limitations of the analysis.

Another practical limitation comes in the form of the typical economic and biophysical models themselves. Landscape-based biophysical models are generally delineated along natural geographic lines, such as a watershed. Economic and social models are often fit to data that is drawn from political boundaries, such as counties or states. An economic model that predicts changes in average cropping practices in a county in response to a price increase is incompatible with a biophysical model based on a watershed. While obvious, this is not a trivial challenge to integration in many cases. On a related note, integrated modeling systems that combine overly simplistic or spatially aggregated models in one component (say, the economic model) with highly detailed and/or spatially disaggregated models in the other area (the biophysical model) will, at best, limit understanding of those systems and, at worst, lead to incorrect conclusions and possibly threaten the value of policy analysis.

Adequate data coverage for model estimation and calibration. A second data challenge facing researchers interested in developing or applying models at the landscape scale is the suitability or appropriateness of adequate data that is consistently collected and available across the entire landscape of interest and available at the appropriate scale. For example, in work reported by Kling et al. (2006), the SWAT model was linked with economic models to predict the costs of adopting a broad set of conservation practices in the Upper Mississippi River Basin and the in-stream water quality effects (reductions in sediment, nitrogen, and phosphorus) from doing so.

While a valuable start on this important problem area, a number of shortcomings related to data availability and quality limit the model's usefulness. First, limited consistent and long-term measurement data on actual in-stream water quality across the basin makes calibration of the parameters in the SWAT model challenging. While data sources exist to accomplish calibration, there is little doubt that more consistent, thorough, and well-documented data would improve the accuracy and believability of model runs.

A second example from this same modeling effort concerned the use and distribution of manure and fertilizer applications on cropped fields in the region. While statewide fertilizer sales are reported, a significant amount of guesswork concerning where this purchased fertilizer is spread in conjunction with where the manure from hog and chicken facilities in the region is applied. While a number of self-reported surveys have been administered and reports from that data are available, there are discrepancies and inconsistencies that make this information hard to rely upon without adjustment to the statewide information.

An example from the economics portion of the integrated model is the availability of cost information on the construction and adoption of conservation practices across multiple states. While Natural Resources Conservation Service (NRCS) offices in each state may collect and report county-level cost-share information for a variety of state and federal conservation programs, those data are not uniformly reported and not available consistently or in a central location. Thus, key information necessary for model building and model calibration are often lacking. Without consistent, long-term data collection to fit and calibrate landscape-level models, the accuracy and usefulness of the models cannot be significantly improved.

Complete model coverage of policy-relevant choices. While data sources may represent a major stumbling point in developing and implementing models, another is a lack in the available modeling capacity itself. Again, drawing from the Upper Mississippi River Basin SWAT-economic model, a major management goal is to reduce sedimentation and movements of phosphorous and other nutrients from farm fields into waterways of the watershed. The SWAT model provides a critically valuable tool on the biophysical side, but does not yet contain methods to site perennial buffers or wetlands within the watershed. Thus, conservation practices, such as reduced tillage, grassed waterways, and land retirement, can all be handled in various modeling scenarios, but a combination of practices with carefully targeted siting of wetlands and buffers cannot. This limits the ability of the modeling paradigm to evaluate many of the policy options that would ideally be considered in planning watershed management for TMDLs in the region or to meet other water quality goals.

In considering the costs of adopting alternative conservation practices, it also is clear that the costs of accepting increased risk in profits and the time costs of farmers belong in the computation of the costs of adopting a particular conservation practice (such as moving from fall to spring fertilizer application). While economists recognize these as appropriate costs and a literature concerning those costs has developed, much is still not understood about the magnitude of the costs and how they vary across farming operations, crops, and locations. Particularly challenging is the cost of nutrient management, which often entails significant learning and time on the part of the farm manager; moreover, there remains a robust debate among agronomists over the short- and long-term yield consequences of reduced nutrient application levels.

Treatment of uncertainty within and across models. A major challenge within the entire scientific community is in reporting and representing uncertainty associated with scientific findings in a way that presents a fair representation of its sources and magnitudes without completely undermining the value of the message. Many of the same issues apply to the case of integrating biophysical and socioeconomic modeling, but here the challenges may be even greater, for two reasons. First, the nature of these models suggest that a great many variables, data sources, and individual models will have to be combined to construct the modeling system and run appropriate scenarios. Each of the variables and models will have various sources of uncertainty associated with them that will, in many cases, be unknown or difficult to quantify.

Second, even if all sources of individual uncertainty could be quantified, many of the sources of uncertainty across models may be correlated. If so, simple addition of weighted standard errors will not yield correct measures of the aggregate variability. Instead, it will be necessary to correct for correlations that can raise or lower the aggregate uncertainty, depending upon the sign of correlation.

Development and implementation of optimization algorithms using models. The challenges associated with integrated modeling do not end once the model is developed. From an economic standpoint, the optimal combination of conservation practices and their location is the spatial configuration of practices that achieves the water quality target at the least possible cost. Unfortunately, as others have recognized (Lintner and Weersink 1999; Khanna et al., 2003), studying the least-cost solution in this context is challenging.

First, the optimal combination may vary dramatically with the chosen level of water quality. That is, the set of conservation practices that will appear in a least-cost solution to achieve a relatively small water quality improvement may differ vastly from the set of practices that appear when a high level of water quality is the objective.

Second, because of the unique nature of the biophysical relationship between conservation practices and resulting water quality levels, the impact of a farmer's conservation practice decision on his field depends also upon the choices of all others' conservation practices, cropping systems, and related decisions in the watershed.

Third, there are multiple abatement possibilities for each field in that many different conservation practices could be implemented. This means that identifying the least-cost solution requires comparing a large number of possible land use scenarios. Specifically, if there are n conservation practices possible for F fields, there will be a total of nF possible configurations. In a watershed with hundreds of fields and eight to ten conservation practices, this comparison quickly becomes unwieldy. An important bright spot in this area is the adaptation of genetic algorithms to watershed models. Those methods were developed for use in general nonlinear and correlated systems, and recent experience suggests they may be useful for approximating the optimal solution to conservation practice placement in a watershed.

Long-term studies and outcome assessment

Comprehensive records are kept on a county basis for all practices installed under USDA programs through the NRCS Performance Results System (PRS). This system tracks most conservation activity on private land in the United States. For most land treatment, the system keeps track of acres treated. For certain practices, such as comprehensive nutrient management planning (CNMP), PRS keeps track of the number of contracts written and the number of acres applied. While the system forms a reasonable basis for estimating cumulative effects of conservation practices, the measurements in PRS are only a first step in understanding the effects of conservation practices. Other measurements are needed to convert PRS data into benefits of conservation practices.

Data on per acre savings in soil, nitrate, and carbon loss and per acre wildlife benefits are also needed. Those estimates are available from only a few studies or modeling efforts. More measurements are needed. Consider that despite billions of dollars invested in the CRP no study has systematically evaluated wildlife response to the CRP from its inception. Presumptive wildlife benefits are inferred from the cumulative weight of evidence from a myriad of site-specific studies (Haufler, 2005). Only recently have wildlife benefits been systematically evaluated on a random sample of contracts for a specific CRP practice (CP33) (Burger et al., 2006).

What is also needed is an understanding of the condition of the practices. Dillaha et al. (1989) studied existing grass filter strips on 18 farms in Virginia and found them to be extremely variable in their effectiveness at removing sediment. Most grass filter strips in hilly areas were ineffective because runoff usually crossed the strip as concentrated flow. In flatter regions, grass filter strips were more effective because slopes were more uniform and more runoff entered the strip as shallow flow. Several one- to three-year-old vegetative filter strips were observed to have trapped so much sediment that they produced more sediment than adjacent upland fields. In those cases, runoff flowed parallel to the vegetative filter strip until it reached a low point, where it crossed the vegetative filter strip as concentrated flow. The vegetative filter strips clearly needed maintenance to regain their sediment-trapping abilities.

These experiences point out the need for tracking not only the application of practices, but the maintenance of practices as well. Annual inspections of practices, such as structural measures (e.g., terraces and grass waterways) and permanent vegetative cover should be possible on all farms receiving cost-share or technical assistance. Other methods, including remote sensing techniques, are needed to understand whether practices, such as residue management and cover crops, are being applied properly. Unfortunately, many of the most important conservation practices, such as nutrient management and pest management, have many aspects that cannot be verified or inspected visually.

Summary and conclusions

The following conclusions have been drawn:

1. Traditional field-scale approaches to environmental research and management have been highly effective in identifying specific field-edge solutions for individual environmental problems. They have been less successful in identifying solutions that are robust across multiple spatial scales, that are socially acceptable, and that meet multiple goals.
2. Needed now are approaches that are systems-oriented (in order to understand direct and indirect outcomes and to balance multiple aims), that are geographically scaleable (to allow effective implementation at watershed and regional scales), that incorporate socioeconomic factors (to allow for the inclusion of human behavior and effective incentives), and that provide for long-term outcome monitoring (to allow solutions to be assessed over time periods that include diverse climatic and other environmental events).
3. To implement those approaches will require resources to provide, in particular, (a) adequate geospatial data coverage, (b) development and deployment of new sensors that

can monitor key environmental attributes over appropriate geographic areas, (c) new modeling approaches that provide for the effective integration of existing biophysical and socioeconomic models, and (d) long-term experiments and outcome assessments.

4. New research, systems-oriented at appropriate geographic and temporal scales, could provide the knowledge needed to develop and implement effective policies for delivering the ecological services that agriculture can provide.

References

Aneja, V. P., B. Y. Wang, D. Q. Tong, H. Kimball, and J. Steger. 2006. Characterization of major chemical components of fine particulate matter in North Carolina. *Journal of the Air & Waste Management Association* 56: 1099-1107.

Beaulieu, J., D. Bennett, S. Kraft, and R. Segupta. 1998. Ecological-economic modeling on a watershed basis: A case study of the Cache River of Southern Illinois. *American Agricultural Economics Association,* Salt Lake City, Utah.

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.

Braden, J. B., G. V. Johnson, A. Bouzaher, and C. Miltz. 1989. Optimal spatial management of agricultural pollutions. *American Journal of Agricultural Economics* 78: 961-971.

Brinson, M. M., R. D. Rheinhardt, F. R. Hauer, L. C. Lee, W. L. Nutter, R. D. Smith, and D. F. Whigham. 1995. A *guidebook for application of hydrogeomorphic assessments to riverine wetlands,* WRP-DE-11. U.S. Army Engineer Waterways Experiment Station, Vicksburg, Mississippi.

Brinson, M. M., and R. D. Rheinhardt. 1996. The role of reference wetlands in functional assessment and mitigation. *Ecological Applications* 6: 69-76.

Burger, L. W. 2005. The Conservation Reserve Program in the Southeast: issues affecting wildlife habitat value. In J. B. Haufler, editor. *Fish and Wildlife Benefits of Farm Bill Conservation Programs:* 2000–2005 Update. Technical Review 05-2. The Wildlife Society Bethesda, Maryland. Pp 63–92.

Burger, L. W. Jr., D. McKenzie, R. Thackston, and S. DeMaso. 2006. The role of farm policy in achieving large-scale conservation: Bobwhite and buffers. *Wildlife Society Bulletin.* 34: 986-993.

Calhoun, F. G., B. K. Slater, and D. B. Baker. 2002. Soils, water quality, and watershed size: interactions in the Maumee and Sandusky River basins of northwestern Ohio. *Journal of Environmental Quality* 31: 47-53.

Carpentier, C. L., D. Bosch, and S. S. Batie. 1998. Using spatial information to reduce costs of controlling agricultural nonpoint source pollution. *Agricultural and Resources Review* 27: 72-84.

Chalfoun, A. D., F. R. Thompson III, and M. J. Ratnaswamy. 2002. Nest predators and fragmentation: a review and meta-analysis. *Conservation Biology* 16:306-318

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.

Clark, W. R. and K. F. Reeder. 2005. Continuous enrollment Conservation Reserve Program: factors influencing the value of agricultural buffers to wildlife conservation. In J. B. Haufler, editor. *Fish and Wildlife Benefits of Farm Bill Conservation Programs:* 2000–2005 Update. Technical Review 05-2. The Wildlife Society, Bethesda, Maryland. Pp. 93 – 114.

Crumpton, W. G. 2000. Using wetlands for water quality improvement in agricultural watersheds: The importance of a watershed scale approach. *Water Science and Technology 44:* 559-564.

Crumpton, W. G. 2005. Potential of wetlands to reduce agricultural nutrient export to water resources in the Corn Belt. *Gulf Hypoxia and Local Water Quality Concerns Workshop.* Iowa State University. Pp. 35-44.

Davie, D. K., and C. L. Lant. 1994. The effect of CRP enrollment on sediment in two Illinois streams. *Journal of Soil and Water Conservation* 49: 407-412.

Dillaha, T. A., J. H. Sherrard, and D. Lee. 1989. Long-term effectiveness of vegetative filter strips. *Water, Environment and Technology* 1: 419-421.

Elmi, A. A., C. Madramootoo, M. Egeh, and C. Hamel. 2004. Water and fertilizer nitrogen management to minimize nitrate pollution form a cropped soil in southwestern Quebec, Canada. *Water, Air, and Soil Pollution* 151: 117-134.

Farrand, D. T. and M. R. Ryan. 2005. Impact of the Conservation Reserve Program on wildlife conservation in the Midwest. In J. B. Haufler, editor. *Fish and Wildlife Benefits of Farm Bill Conservation Programs: 2000–2005 Update.* Technical Review 05-2. The Wildlife Society Bethesda, Maryland. Pp. 41–62.

Feng, H., L. Kurkalova, C. Kling, and P. Gassman. 2006. Environmental conservation in agriculture: Land retirement vs. changing practices on working land. *Journal of Environmental Economics and Management* 52: 600-614.

Feng, H., L. Kurkalova, C. Kling, and P. W. Gassman. Economic and environmental co-benefits of carbon sequestration in agricultural soils: Retiring agricultural land in the Upper Mississippi River Basin. *Climatic Change* (in review).

Ferriter, A. 1997. Brazilian pepper management plan for Florida. Florida Exotic Plant Council.

Findlay, S. E., E. Kiviat, W. C. Nieder, and E. A. Blair. 2002. Functional assessment of a feference wetland set as a tool for science, management, and restoration. *Aquatic Sciences* 64: 107-117.

Florinsky, I. V. 1998. Accuracy of local topographic variables derived from digital elevation models. *International Journal of Geographical Information Science* 12: 47-61.

Gallant, J. C., and J. P. Wilson. 2000. Primary topographic attributes in J. P. Wilson and J. C. Gallant, eds. *Terrain Analysis: Principals and Applications.* John Wiley and Sons, Inc., New York, New York.

Gburek, W. J., C. C. Drungil, M. S. Srinivasan, B. A. Needleman, and D. E. Woodward. 2002. Variable source area controls on phosphorus transport: bridging the gap between research and design. *Journal of Soil and Water Conservation* 57: 534-543.

Haufler, J. B., editor. 2005. *Fish and Wildlife Benefits of Farm Bill Conservation Programs:* 2000–2005 Update. Technical Review 05-02. The Wildlife Society, Bethesda, Maryland.

Horn, D. J., M. L. Phillips, R. R. Koford, W. R. Clark, M. A. Sovada, and R. J. Greenwood. 2005. Landscape composition, patch size, and distance to edges: Interactions affecting duck reproductive success. *Ecological Applications* 15: 1367-1376.

Intergovernmental Panel on Climate Change. 2002. *IPCC Special Report on Land Use, Land-Use Change and Forestry.* Cambridge University Press, Cambridge, United Kingdom.

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

Johansson, R. C., P. H. Gowda, D. J. Mulla, and B. J. Dalzell. 2004. Metamodeling phosphorus best management practices for policy use: A frontier approach. *American Journal of Agricultural Economics* 30: 63-74.

Johnson, D. H. 2005. Grassland bird use of Conservation Reserve Program fields in the Great Plains. In J. B. Haufler, editor. *Fish and Wildlife Benefits of Farm Bill Conservation Programs: 2000–2005 Update.* Technical Review 05-02. The Wildlife Society, Bethesda, Maryland. Pp. 17–32.

Khanna, M., W. Yang, R. Farnsworth, and H. Onal. 2003. Cost-effective targeting of land retirement to improve water quality with endogenous sediment deposition coefficients. *American Journal of Agricultural Economics* 85: 538-553.

Kling, C., S. Secchi, M. Jha, H. Feng, P. W. Gassman, and L. A. Kurkalova. 2006. Upper Mississippi River Basin modeling system part 3: Conservation practice scenario results. In V. P. Singh and Y. J. Xu, editors, *Coastal Hydrology and Processes.* Water Resources Publications, Highland Ranch, Colorado.

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

Kurkalova, L., C. Kling, and J. Zhao. 2006. Green subsidies in agriculture: estimating the adoption costs of conservation tillage from observed behavior. *Canadian Journal of Agricultural Economics* 54: 247-267.

Landis, D. A., F. D. Menalled, A. C. Costamagna, and T. K. Wilkinson. 2005. Manipulating plant resources to enhance beneficial arthropods in agricultural landscapes. *Weed Science* 53: 902-908.

Lintner, A. M., and A. Weersink. 1999. Endogenous transport coefficients. *Environmental and Resource Economics* 14: 269-296.

Lowrance, R. R., R. L. Todd, and L. E. Asmussen. 1984. Nutrient cycling in an agricultural watershed. II. Streamflow and artificial drainage. *Journal of Environmental Quality* 13: 27-32.

Lowrance, R. 1998. Riparian forest ecosystems as filters for nonpoint-source pollution. Pages 113-141 in M. L. Pace and P. M. Groffman, eds. *Successes, Limitations, and Frontiers in Ecosystem Science.* Springer Verlag, New York, New York.

Mausbach, M. J., and A. R. Dedrick. 2004. The length we go: measuring environmental benefits of conservation practices. *Journal of Soil and Water Conservation* 59: 96-103.

Moore, I. D., R. B. Grayson, and A. R. Ladson. 1991. Digital terrain modeling: a review of hydrological, geomorphological and biological applications. *Hydrology Proceedings* 5: 3-30.

Moore, I. D., P. E. Gessler, G. A. Nielsen, and G. A. Peterson. 1993. Soil attribute prediction using terrain analysis. *Soil Science Society of America Journal* 57: 443-452.

National Research Council. 2000. Clean coastal waters: Understanding and reducing the effects of nutrient pollution. National Research Council, National Academy Press, Washington, D.C.

National Research Council. 2003. Frontiers in agricultural research. Food, health, environment, and communities. National Academy Press, Washington, D.C, USA.

Petrolia, D., P. H. Gowda, and D. J. Mulla. 2005. Targeting agricultural drainage to reduce nitrogen losses in a Minnesota watershed. Staff Paper Series P-05-2. Department of Applied Economics, University of Minnesota. http://agecon.lib.umn.edu/mn/p05-02.pdf.

Phillips, M. L., W. R. Clark, S. M. Nusser, M. A. Sovada, and R. J. Greenwood. 2004. Analysis of predator movement in prairie landscapes with contrasting grassland composition. *Journal of Mammalogy* 85:187 – 195.

Phillips, M. L., W. R. Clark, M. A Sovada, D. J. Horn, R. R. Koford, and R. J. Greenwood. 2003. Predator selection of prairie landscape features and its relation to duck nest success. *Journal of Wildlife Management* 67:104 – 114.

Qiu, Z., and T. Prato. 1999. Accounting for spatial characteristics of watersheds in evaluating water pollution abatement policies. *Journal of Agricultural and Applied Economics* 31: 161-175.

Randall, G. W., and J. E. Sawyer. 2005. Nitrogen application timing, forms, and additives. Gulf Hypoxia and Local Water Quality Concerns Workshop. Iowa State University, Ames, Iowa. Pp. 73-84.

Reynolds, R. E. 2005. The Conservation Reserve Program and duck production in the U.S. Prairie Pothole Region. In J. B. Haufler, editor, Fish and Wildlife Benefits of Farm Bill Conservation Programs: 2000–2005 Update. Technical Review 05-2. The Wildlife Society Bethesda, Maryland. Pp. 17–32.

Ribaudo, M., R. Heimlich, R. Claasen, and M. Peters. 2001. Least-cost management of nonpoint source pollution: source reduction versus interception strategies for controlling nitrogen loss in the Mississippi basin. *Ecological Economics* 37: 183-197.

Richards, R. P., and D. B. Baker. 2002. Trends in water quality in LEASQ rivers and streams (Northwestern Ohio), 1975-1995. *Journal of Environmental Quality* 31: 90-96.

Richardson, C. J., and S. S. Qian. 1999. Long-term phosphorus assimilative capacity in freshwater wetlands: A new paradigm for sustaining ecosystem structure and function. *Environmental Science & Technology* 33: 1545-1551.

Riffell, S. K. and L. W. Burger. 2006. Estimating Wildlife Response to the Conservation Reserve Program: Bobwhite and Grassland Birds. Final report for FSA-R-28-04DC Estimating Wildlife Response to the Conservation Reserve Program, Department of Agriculture, Farm Service Agency, Acquisition Management Branch, Special Projects Section. http://www.fsa.usda.gov/Internet/FSA_File/quail_study.pdf (accessed 12/03/06).

Robertson, G. P. 2004. Abatement of nitrous oxide, methane, and the other non-CO2 greenhouse gases: The need for a systems approach. In C. B. Field and M. R. Raupach, editors, The Global Carbon Cycle. Island Press, Washington, D.C. Pp. 493-506.

Robertson, G. P., and S. M. Swinton. 2005. Reconciling agricultural productivity and environmental integrity: A grand challenge for agriculture. *Frontiers in Ecology and the Environment* 3: 38-46.

Robertson, G. P., J. C. Broome, E. A. Chornesky, J. R. Frankenberger, P. Johnson, M. Lipson, J. A. Miranowski, E. D. Owens, D. Pimentel, and L. A. Thrupp. 2004. Rethinking the vision for environmental research in U.S. agriculture. *BioScience* 54: 61-65.

Russell, C. S., and J. F. Shogren. 1993. Theory, modeling and experience in the management of nonpoint-source pollution. Kluwer Academic Publishers.

Ryan, M. R. 2000. Impact of the Conservation Reserve Program on wildlife conservation in the Midwest. In W. L. Hohman and D. J. Halloum, editors, A Comprehensive Review of Farm Bill Contributions to Wildlife Conservation, 1985–2000. Technical Report USDA/NRCS/WHMI-2000. U.S. Department of Agriculture, Forest Service, Washington, D.C. Pp. 45–54.

Soil and Water Conservation Society. 2006. Final report from the Blue Ribbon Panel conducting an external review of the U.S. Department of Agriculture Conservation Effects Assessment Project. Ankeny, Iowa.

Stephens, S. E., D. N. Koonsa, J. J. Rotella, and D. W. Willey. 2003. Effects of habitat fragmentation on avian nesting success: a review of the evidence at multiple spatial scales. *Biological Conservation* 115: 101–110.

Sunding, D., and D. Zilberman. 2000. The agricultural innovation process: research and technology adoption in a changing agricultural sector. In B. Gardner and G. Rausser, editors, Handbook of Agricultural Economics. Elsevier Science Publishers.

Swinton, S. M., F. Lupi, G. P. Robertson, and D. A. Landis. 2006. Ecosystem Services from Agriculture: Looking Beyond the Usual Suspects. *American Journal of Agricultural Economics* 88:1160-1166

Tan, C. S., C. F. Drury, W. D. Reynolds, P. H. Groenvelt, and H. Dalfar. 2002. Water and nitrate loss through tiles under a clay loam soil in Ontario after 42 years of consistent fertilization and crop rotation. *Agriculture, Ecosystems & Environment* 93: 121-130.

Theobald, D. M. 1989. Accuracy and bias issues in susfrace represemtation. In M. Goodchild and S. Gopol, editors, Accuracy of Spatial Databases. Taylor and Francis, Bristol, Pennsylvania. Pp. 99-106.

Thompson, F. R. III, T. M. Donovan, R. M. DeGraaf, J. Faaborg, and S. Robinson. 2002. A multi-scale perspective on the effects of forest fragmentation on birds in eastern forests. *Studies in Avian Biology* 25:8 – 19.

Twedt, D. J. and W. B. Uihlein III. 2005. Landscape level reforestation priorities for forest breeding landbirds in the Mississippi Alluvial Valley. In L. H. Fredrickson, S. A. King, and R. M. Kaminski, editors, Ecology and Management of Bottomland Hardwood Systems: The State of our Understanding. Gaylord Memorial Laboratory Special Publication No. 10, Puxico.University of Missouri-Columbia. Pp. 321-340.

Udawatta, R. P., J. J. Krstansky, G. S. Henderson, and H. E. Garrett. 2002. Agroforestry practices, runoff, and nutrient loss: A paired watershed comparison. *Journal of Environmental Quality* 31: 1214-1225.

Vellidis, G., R. Lowrance, P. Gay, R. W. Hill, and R. K. Hubbard. 2003. Nutrient transport in a restored riparian wetland. *Journal of Environmental Quality* 32: 711-726.

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 and Research* 30: 49-74.

Wilson, A. T. 1978. The explosion of pioneer agriculture: contribution to the global CO2 increase. *Nature* 273: 40-41.

Zedler, J. B., and S. Kercher. 2004. Causes and consequences of invasive plants in wetlands: Opportunities, opportunists, and outcomes. *Critical Reviews in Plant Science* 23: 431-452.

Roundtable:
Methods for environmental management

At the outset of the roundtable, participants reiterated two important goals of the U.S. Department of Agriculture's Conservation Effects Assessment Project: (1) Estimate the effects of conservation practices and (2) expand those estimates for the entire nation. Discussion then focused on several important talking points:

- Do we need standardized research methods? In that practical given different ecoregions, microclimates, geomorphology, and so forth?
- Do we have a consensus on the "state of the art" in terms of methods and approaches? Do we know what everyone is doing?
- Are we gathering the right types of data? Are those types of data what the modelers will need?
- Are our methods suitably precise?
- Do we agree on what our practices are supposed to be doing, for example, reducing sediment, enhancing wildlife, processing nutrients?
- Using our methods, can we detect confounding effects or conflicting effects? For example, one conflicting effect: Winter cover reduces sediment, but may increase organic loading, which may reduce dissolved oxygen. One confounding effect: Reduced sediment increases light, which increases chlorophyll a, but reducing sediment also reduces phosphorus, which reduces chlorophyll a.

Roundtable participants then identified the six most important "next steps" necessary to strengthen the science associated with environmental management research methods or approaches:

1. Better understanding of current model limitations and development of models that account for larger scale processes (stream channel processes, reaction kinetics). If important processes are not adequately represented, calibration may match results for the wrong reasons, and extrapolation to changed conditions will not reflect real results.
2. Long-term (30 or more years) geospatial datasets suitable for calibrating and testing the next generation of models (long-term geospatial data at different scales, including waterways). Get major players (U.S. Geological Survey, U.S. Environmental Protection Agency, U.S. Department of Agriculture,…) to work together to populate and maintain a data network.
3. Standardize methodologies for data collection, sampling protocols, and analytical procedures. Prepare quality control and assurance procedures.
4. Standardization and/or development of a methodology to declare "significant differences or effects." Techniques for analyzing uncontrollable, unreplicable situations. Error estimates, variance components, meta-analysis, time series. What are the acceptable measures to declare that "we did or did not make a difference"?
5. Identify specific experiments that need to be done across all watershed studies as work moves from the field level to the watershed level. Provide the infrastructure and support to conduct experiments of processes at multiple spatial and temporal scales (like long-term ecological research, multiple paired watersheds, and so forth).
6. Develop tools to tailor adaptive management to an agricultural watershed context.