



Table of Contents

2nd Quarter 2007—Volume 22, Number 2

- A Statement from the Editors and AAEA President 75
Washington Scene 77
 Coordinated by Joe L. Outlaw, Co-Editor, Choices

Articles

Theme: Resources and the Environment

Water Quality and Agriculture *(Catherine Kling, Guest Editor)*

- Overview: Agriculture and Water Quality in the Cornbelt: Overview of Issues and Approaches 79
 Matthew J. Helmers, Thomas M. Isenhardt, Catherine L. Kling (Guest Editor), Thomas B. Moorman, William W. Simpkins, and Mark Tomer
- A Tale of Three Watersheds: Nonpoint Source Pollution and Conservation Practices Across Iowa 87
 Keith E. Schilling, Mark D. Tomer, Philip W. Gassman, Cathy L. Kling, Thomas M. Isenhardt, Thomas B. Moorman, William W. Simpkins, and Calvin F. Wolter
- Privatizing Ecosystem Services: Water Quality Effects from a Carbon Market 97
 Silvia Secchi, Manoj Jha, Lyubov Kurkalova, Hongli Feng, Philip Gassman, and Catherine L. Kling
- Nitrate Reduction Approaches 103
 Christopher Burkart and Manoj K. Jha

Grab Bag

- Organic Demand: A Profile of Consumers in the Fresh Produce Market 109
 John Stevens-Garmon, Chung L. Huang, and Biing-Hwan Lin
- Water Quality Credit Trading and Agriculture: Recognizing the Challenges and Policy Issues Ahead. 117
 Charles Abdalla, Tatiana Borisova, Doug Parker, and Kristen Saacke Blunk
- Farm Growth, Consolidation, and Diversification: Washington Dairy Industry 125
 Tristan D. Skolrud, Erik O'Donoghue, C. Richard Shumway, and Almuhanad Melhim
- Fruit and Vegetables Go Back to School 129
 John L. Park, Benjamin L. Campbell, Andres Silva, and Rodolfo M. Nayga, Jr.



A Statement from the Editors and AAEA President

Welcome to the twelfth and final issue of our editorship of *Choices* (Q2 2007). As discussed in the AAEA President's statement just below, the Association intends to continue *Choices* and hopes to have it back online by the end of 2007. Watch for announcements from the AAEA later this year.

Our term as editors expires with this issue. We wish to thank those who have served on the editorial board as well as those who have served as reviewers during our editorial term. Special thanks are due to those individuals who served as guest editors for specific issues. *Choices* had a fantastic run the last 3 years as an outreach vehicle for the association. Thank you for your interest.

As our final offering, this issue contains a theme on water quality in the Cornbelt dealing with the problem origin, and issues regarding conservation programs, multiple service provisions and tradeoffs and water treatment options. This issue also contains articles on

- Organic Produce Consumer Characteristics
- Challenges in Water Quality Credit Trading in Agriculture
- Dairy Farm Growth, Consolidation, and Diversification
- Fruit and Vegetables in School Food

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Choices is the outreach vehicle of the American Agricultural Economics Association (AAEA) and is designed to provide current coverage regarding economic implications of food, farm, resource, or rural community issues directed toward a broad audience. *Choices* publishes thematic-oriented groupings of papers and individual papers. The broad themes we will repeatedly visit in *Choices* are agriculture and trade, resources and the environment, consumers and markets, and agribusiness and finance. Submitted manuscripts are subject to peer review for publication consideration.

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Editorial Communications

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Again thank you for your readership and participation.

Co-Editors: Oral Capps, Jr., Rodolfo Nayga, Jr., Bruce McCarl, Joe Outlaw, and John B. Penson, Jr., and Associate Editor, Linda Crenwelge

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A Message from AAEA President Steve Buccola

CHOICES has been the American Agricultural Economics Association's outreach arm for 23 years, our principal means of communicating with those interested in food, farm, natural resource, and rural community issues but who are not necessarily

professional economists. *CHOICES* brings economic research alive to a policy audience, and internet hits and downloads suggests it has been increasingly successful in doing so. Policy communication is essential to the AAEA's ethic because it is essential to the mission to which many of our members – and their employers – are committed.

CHOICES has always run at a substantial financial loss. AAEA members have, in constant 2006 dollars, contributed \$1,262,018 to this journal (\$21 per member per year) since 1990 alone. Annual losses were reduced when *CHOICES* went electronic but still have hovered around \$50,000 and, because our membership has declined, is still \$20 per member per year. The financial environment in which the AAEA oper-

ates requires that it further reduce these costs by providing *CHOICES* with a new publishing and editorial structure. Plans for doing so are well underway.

The AAEA is deeply grateful to the editorial team of Bruce McCarl (Coordinating Editor), Oral Capps, Jr., Rodolfo Nayga, Jr., Joe Outlaw, and John B. Penson, Jr. at Texas A&M University, to Associate Editor Linda Crenwelge, and to the 17-member Editorial Board for raising *CHOICES* to new standards of clarity, relevance, and accessibility. Their work provides a bridge to our next format. Stay tuned.

Steve Buccola

President, American Agricultural Economics Association



Washington Scene

Coordinated by Joe L. Outlaw, Co-Editor, *Choices*

Congress is currently trying to wrap up loose ends on several key pieces of legislation before their summer recess, which encompasses most of August and the early part of September. The Farm Bill, a new Energy Bill, and immigration reform, among other issues such as appropriations bills, have been on their agenda.

Farm Bill

Thus far, the House and Senate Agricultural Committees are each moving forward at their own pace. The House has held subcommittee hearings to debate almost all of the titles of the Farm Bill. The Subcommittee on General Commodities and Risk Management voted 18-0 to use the 2002 Farm Bill as their bill for Title I. That doesn't mean there won't be changes in full committee, but that at least the subcommittee intends for the basic structure of the 2007 Farm Bill to look much like the 2002 Farm Bill. Full committee markup in the House has been postponed, but would still happen in time for the scheduled debate in the entire House of Representatives in late July.

Mr. Harkin, the Senate Agriculture Committee Chair, has been working through all of the requests from interest groups to develop his proposal (generally referred to as the Chairman's mark). To date, there has not been any floor time scheduled to debate the farm bill in the Senate. Observers of the farm bill process over the past 20 years will say that the Senate tends to move through the process slower than the House of Representatives, so a slower pace isn't unexpected.

When a bill is finally passed, what is it going to look like? At this point, most farm policy observers would say that either an extension of the 2002 Farm Bill or a slightly modified 2002 Farm Bill are the two most likely outcomes for Title I. Does this mean that there wouldn't be any changes? No, there will likely be modest changes in com-

modity programs, along with increased financial support in many areas such as food, conservation, and energy programs to name a few. However, the wholesale changes in the commodity program are less likely than some want because there isn't enough money available to change the commodity programs and provide a safety net that works as well as the current one.

Doha Round

Only a week after WTO Director-General Pascal Lamy reported modestly high hopes for gaining an agreement within the next six months, negotiators from the G4 (European Union, United States, Brazil, and India) failed to move forward at their meeting in Potsdam. What next? Trade Promotion Authority (TPA) expires on June 30th. Based on comments from members of Congress, it isn't clear whether a Doha Round agreement would have survived the up or down vote with TPA. Without TPA, the chances of the Congress passing (without amendment) any Doha Round agreement seem highly unlikely.

Energy

The Senate has been debating a new energy bill (HR 6) that was passed on June 22nd. In order to gain passage, a compromise was made on corporate average fuel economy (CAFE) standard increases – lowering the mandated increase to a fleetwide average of 35 miles per gallon. The bill would mandate 36 billion gallons of renewable fuel by 2022, increase efficiency standards for appliances and federal buildings, promote new energy technologies, and provide federal grants and loan guarantees for research into fuel-efficient vehicles.

The House has not cleared their version of the Energy Bill.



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

By Matthew J. Helmers, Thomas M. Isenhart, Catherine L. Kling, Thomas B. Moorman, William W. Simpkins, and Mark Tomer

JEL Classification Code: Q25

More than three decades have elapsed since the passage of the Federal Water Pollution Control Act with its stated goal of zero discharge of pollutants into the nation's waterways. Yet, water quality remains poor in many locations and considerable loading of pollutants continues. This is particularly true for agricultural sources of water pollution and is typified by the Upper Mississippi River Basin, where more than 1,200 water bodies appear on the current U.S. Environmental Protection Agency (EPA) listing of impaired waterways. Additionally, nitrate export from this region has been implicated as a significant cause of the hypoxic zone in the Gulf of Mexico, which covered nearly 20,000 km² in 1999 and more than 17,000 km² in 2006 (http://www.epa.gov/gmpo/nutrient/hypoxia_pressrelease.html). Although a substantial body of evidence on the effectiveness of agricultural conservation practices on water quality continues to be developed, the net effect of these programs and practices at the watershed scale is unclear. Increasingly, studies are being focused on the watershed (or landscape) scale and complex interactions between agricultural practices and inputs, the types and configuration of conservation practices on the landscape, and the resulting downstream water quality. While low cost methods to reduce agricultural non-point source pollution exist, large changes in water quality in agricultural regions are likely to be costly and met with resistance. This is because to achieve large changes in water quality, major alterations to land use or installation of expensive struc-

tural practices may be required, and the costs are borne directly by producers and landowners, or by the taxpayer.

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

In the third paper of the series, Secchi et al. employ a more aggregate unit of analysis (scale) for calibrating a watershed model and a biophysical carbon sequestration model and integrating them with economic data covering the entire state of Iowa. The focus of their analysis is on

the potential unanticipated environmental effects of developing markets in ecosystem services that focus on a single service, such as carbon sequestration.

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

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

Agriculture and Water Quality Primer

Production of food and fiber have inevitable impacts on land and water resources. Conservation practices are

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

Nitrates-Nitrogen

Nitrate-nitrogen ($\text{NO}_3\text{-N}$) is a key pollutant of concern for its potential widespread impact on both public health and ecosystem function. Nitrate-nitrogen is readily leached through soils to groundwater and enters surface water systems directly by groundwater flow and through the subsurface drainage systems (tile drains), which were installed across large areas of poorly drained Midwestern soils beginning about 100 years ago. These drainage systems have allowed the Midwest to become the highly productive agricultural area that it is today, while short-circuiting the much slower, natural groundwater pathway to the stream. Concentrations of nitrate-nitrogen in drainage and stream water often exceed 10 mg $\text{NO}_3\text{-N} / \text{L}$, resulting

in losses exceeding 20 kg N/ha in some years (Tomer et al., 2003). Regional nitrogen budgets for the Mississippi River Basin have implicated tile-drained regions of the Midwest as disproportionately contributing to N loads to the Gulf (Burkart and James, 1999). Nitrogen fertilizer is commonly applied to corn, at rates varying from 100 to 200 kg/ha. The efficiency of N uptake by the crop varies because of environmental conditions. Nitrogen losses are most prevalent in early Spring when crops are not present or are too small to effectively immobilize the available nitrate.

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

Phosphorous and Sediment

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

Sediment and nutrient losses from agriculture, therefore, can result in a legacy of impacts within watersheds, necessitating a long-term commitment to their amelioration. For

example, elevated nitrate concentrations in groundwater have been shown to remain for decades (Rodvang and Simpkins, 2001). Also, phosphorus accumulations in sediment may have a legacy, providing a long-term, internal loading source of mineral P to the water column (Christophoridis and Fytianos, 2006) and may ultimately affect groundwater P concentrations (Burkart et al., 2004).

Bacterial Pathogens and Livestock Concerns

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

loss of excess nutrients (Kellogg et al., 2000).

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

Heterogeneity of Conservation Practices

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

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

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

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

We have also learned that some agricultural practices have effects that were not intended. Subsurface drain-

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

While there is a wide range of practices that can be used on agricultural lands for providing water quality benefits, many times the locations within the watershed where practices are implemented have not been specifically targeted to achieve the greatest reduction of contaminants in downstream water bodies. This is likely the result of the voluntary enrollment in federal conservation programs combined with ineffective targeting technology. Recent advances in remote sensing and geographic information systems offer an opportunity for dramatic improvements in our ability to target conservation practice installation in large watersheds. With the limited amount of resources available for conservation practices, there will likely be increased importance on targeting implementation to those areas where

there may be the greatest benefit from a water quality perspective. One program that has used targeting with some effectiveness is the Conservation Reserve Program (CRP), which targets land choices based on an environmental benefits index. While the effects of CRP on soil quality, carbon storage, and wildlife have been assessed, the aggregate effects at the watershed scale are less understood.

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

The Policy Environment: TMDLs and Voluntary Implementation

Voluntary cost-share and incentive programs sponsored by USDA and States are large in geographic scale and fiscal commitment (over \$4.5 billion was spent in 2005 by USDA-funded programs alone). These programs generally provide varying incentives to farmers for the installation of structural or management practices described above. The criteria for participant eligibility vary from program to program, and conservation compliance provisions require that landowners who farm on highly erodible land undertake some conservation activities in order to be eligible for other government incentives or subsidies. In addition to the largest program, the CRP, there is a cost-share program entitled the Environmental Quality Incentive Program, which provides cost share to producers willing to install various conservation structures or practices

on their farms. Notably, the 2002 Farm Bill contained a new program—the Conservation Security Program—a watershed-based initiative intended to compensate farmers for adopting conservation practices. Like the CRP, which covers the full cost of retiring land from production, the program was intended to cover the full cost of adopting conservation practices (rather than less than 100% of the cost as traditional cost share programs do), but the focus of the Conservation Security Program is on land that stays in production. However, funding constraints have prevented the program as it was initially envisaged from being fully implemented.

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

because of microbial pollution (Smith & Perdek, 2004).

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

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

In the TMDL process, modeling and monitoring can play important roles in allocating pollutant loads to various sources, such as helping to determine the relative contributions of row crops, CAFOs, and urban sources to loads of nutrients and bacteria observed in large watersheds. Two models, the Soil and Water Assessment Tool and the Hydrological Simulation Program-FORTRAN models are most often used to support TMDL assessments (Benham et al., 2006). These models combine GIS-based spatial data of watershed physical features with information on cropping systems, animal densities, fertilizer and pesticide use, and point

sources. For non-point source pollutants, conservation practices are a key to developing mitigation strategies that allow watersheds to meet TMDL goals. Since TMDLs may be designed to mitigate multiple pollutants (e.g., nitrate and bacteria), combinations of conservation practices may be necessary to achieve the necessary improvements in water quality.

Final Remarks

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

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

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

Before we leave the reader to dive into the three following papers, we note a final thorny point concerning the potential for significant "legacy" problems possibly hiding in groundwater supplies. Over many decades of agricultural activity, we have added nutrients and other effluents to

groundwater systems that have undoubtedly not yet emerged at the surface. When and where such pollutants will appear is not clear, but if conservation programs are designed only with current pollutant contributions in mind, our efforts may well fall short due to these legacy sources.

For More Information

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

By Keith E. Schilling, Mark D. Tomer, Philip W. Gassman, Cathy L. Kling, Thomas M. Isenhart, Thomas B. Moorman, William W. Simpkins, and Calvin F. Wolter

JEL Classification Code: Q25

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

Within agricultural regions, one might expect greater homogeneity in biophysical features and cropping practices and be tempted to think that one size fits all; i.e., that one set of conservation prescriptions can be used to address the negative impacts of agriculture on aquatic and terrestrial integrity. If this generalization could be made anywhere, certainly a state such as Iowa, dominated by its vast extent of corn and soybean fields, would be suited for a limited set of conservation prescriptions. However, as

described in this tale of three watersheds, conservation practices must instead be tailored to individual landowner objectives and local landscape conditions in order to optimize their effectiveness.

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

Watershed Descriptions

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

Landforms

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

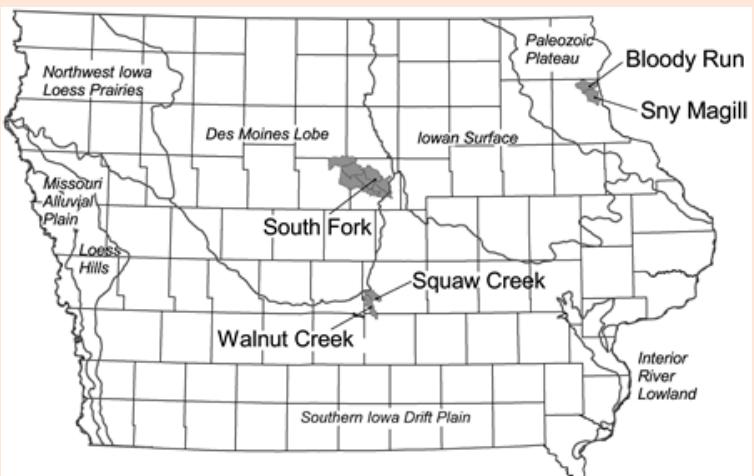


Figure 1. Location of the three watersheds (and controls) and EPA Ecoregions in Iowa.



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

1991). In Northeast Iowa, Sny Magill Creek is a third-order stream in Clayton County that drains 35.6 mi² of the Paleozoic Plateau landform region before discharging directly into the Mississippi River. The landscape of this region is characterized by narrow, gently sloping uplands that break into steep slopes with abundant outcrops of sandstone and limestone. The characteristic limestone bedrock of the area gives rise to karst features (sinkholes, caves, and springs) that are found throughout the Sny Magill Creek watershed (Figure 2).

Nearly 80% of annual streamflow is ‘baseflow’ attributable to ground water discharge from these subsurface sources. This results in “cold water” conditions suitable for highly popular trout fisheries.

The 18.3 mi² Squaw Creek watershed is located in South-Central Iowa in Jasper County in the Southern Iowa Drift Plain. The landscape of this region is characterized by steeply rolling hills and a well-developed stream network that developed on a landscape composed of geologically old (>500,000 years) glacial till

(poorly sorted mixture of gravel, sand, silt, and clay) overlain by geologically recent (17,000 to 31,000 year old) windblown silt (loess). Because of the sloping hillsides and poor infiltration capacity of the soils, rainfall is primarily directed to streams via overland runoff, and only 55% of the stream discharge is attributable to baseflow originating as ground water.

The largest of the study watersheds is the South Fork of the Iowa River, which covers 301 mi² within Hardin and Hamilton counties in Central Iowa. The landscape is representative of the Des Moines Lobe, the dominant landform region of North-Central Iowa. The terrain is young (about 12,000 years since glacial retreat), and thus much of the landscape is dominated by low relief and poor surface drainage. Prior to settlement by Europeans, the landscape was a complex of wetlands, and the stream network was poorly developed due to the relatively young landscape. The geology of the Des Moines Lobe region consists largely of glacial till deposits in moraines and flat to rolling uplands, clay and peat in depressional “prairie pothole” areas, and sand and gravel deposits in floodplains of rivers and streams. Soil wetness is a major concern for land management and agricultural production. Hydric soils (indicative of soil saturation on at least a seasonal basis) occupy about 54% of the watershed, and artificial tile drainage (Figure 2) was installed in these highly productive and nutrient rich soils to lower the water table and allow crops to be grown. Thus, about 70% of the stream flow in the South Fork watershed originates from subsurface drainage (Green et al., 2006), with most tile discharge occurring during spring and early summer.

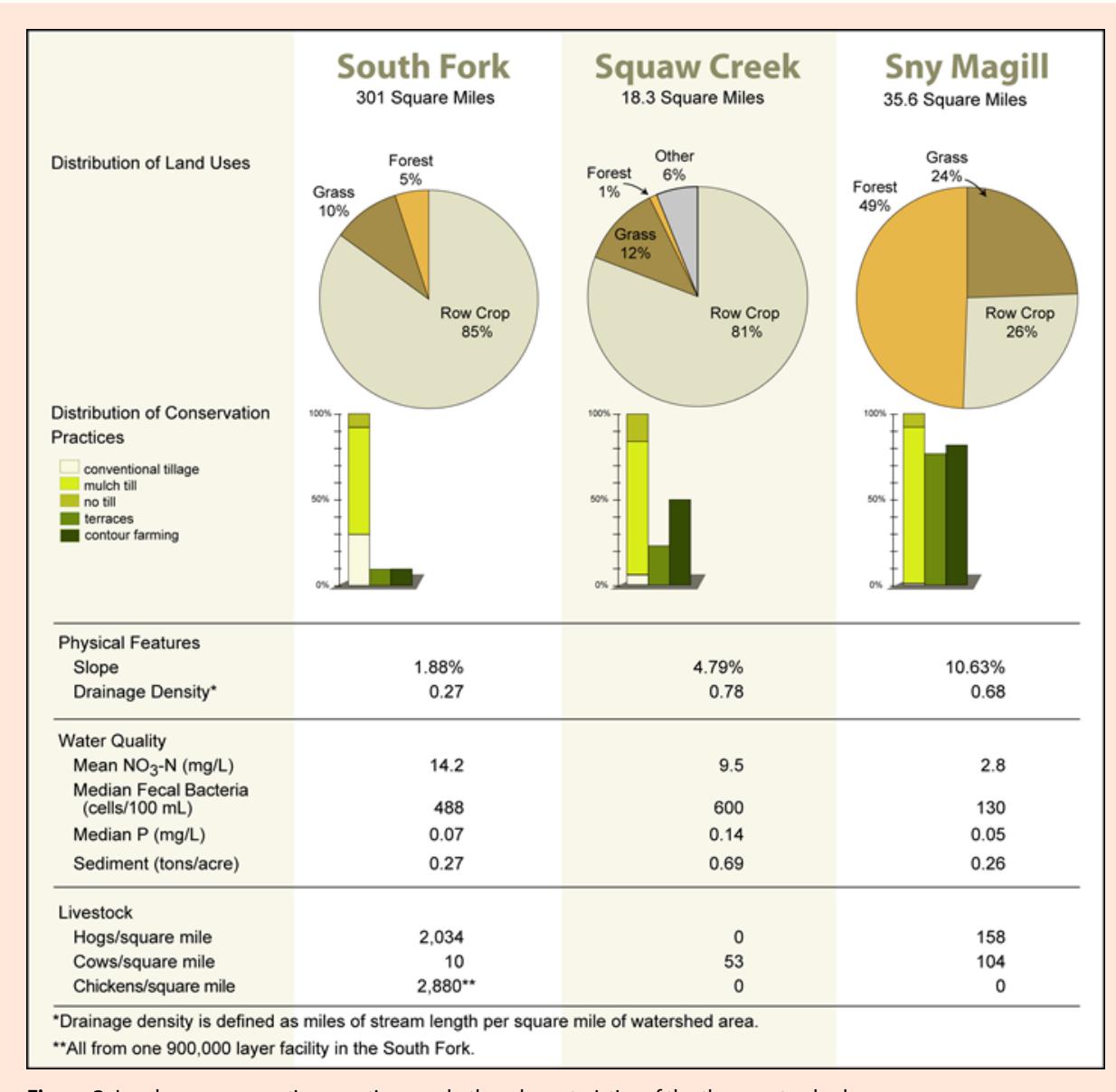


Figure 3. Land use, conservation practices, and other characteristics of the three watersheds.

Physical features of the three watersheds have a great influence on the timing and magnitude of the routing of water to streams. Watersheds draining older landscapes have greater slope and greater stream density (number of streams per square mile) than younger landscapes (Figure 3). For example, the Sny Magill watershed has twice the average slope as Squaw Creek, which has more than twice the slope of the South Fork watershed. Slopes in Sny Magill are further accentuated because of

the bedrock terrain and its proximity to the Mississippi River. The Squaw Creek and Sny Magill watersheds also have nearly three times more streams per square mile than the South Fork. The well-dissected landscape of the Sny Magill watershed shows a greater stream density; thus, rainfall can be quickly routed as overland runoff to sinkholes or streams. In the South Fork watershed, where natural drainage is poor, excess rainfall would collect in potholes or other surface depressions if not for prevalence of

subsurface tile drainage, which has accelerated the routing of rainfall water off the land. Watersheds draining the Des Moines Lobe may yield as much water as those draining fractured carbonate bedrock (Schilling and Wolter, 2005).

Relation of Land Use to the Landform Region

Differences in land cover among the three watersheds can be traced largely to their watershed morphologies and the suitability of land for intensive

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

Animal Agriculture

In the early 1900s, most small farms in Iowa had livestock, often including cattle, swine, and chickens. As a result of changes in farm policy and economies of scale, all three watersheds have experienced shifts in animal agriculture that are representative of changes across the larger landform regions (Figure 1). Historically, the Sny Magill watershed had significant numbers of dairy cattle utilizing available grasslands for forage. While still a significant industry, dairy cattle within Sny Magill have decreased greatly, with a resulting shift in some grassland acreage to row crop agriculture (soybean acreage especially increased in Iowa as pasture and hayland decreased). Livestock is comparatively absent in the Squaw Creek watershed except for several cow-calf operations. Nowhere is the concentration of livestock more apparent than in the South Fork watershed, where most swine and chickens are raised in confined animal feeding operations (CAFOs).

There are nearly 100 CAFOs (mostly swine) in the watershed. Based on inventories reported for permitted facilities, hogs and chickens in the South Fork watershed number 1,654 and 2,880 per square mile, respectively, which are densities considerably greater than the other two watersheds combined (Figure 3). All the reported chickens are housed in one large egg-producing facility, while swine facilities are abundant across the central part of the watershed. We estimate that about a quarter of the watershed receives manure applications annually, assuming this is applied prior to corn at a rate equivalent to that crop's uptake of nitrogen (about 190 lb N/ac). Usually these applications are done by injection, and carried out in the fall when soils tend to be dry and most easily trafficked by manure tankers and applicators.

NPS Pollutants and the Landscape

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

the highest observed in the United States (Becher et al., 2001).

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

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

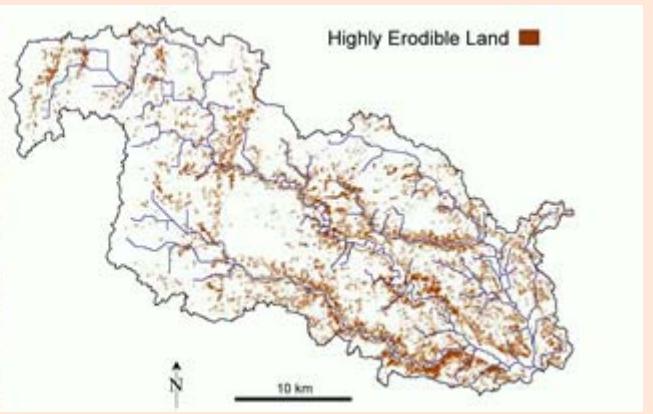


Figure 4. Eroding streambanks and erodible soils are possible contributors to sediment loads. In the South Fork watershed, both cut-bank meanders and erodible soils are mostly found in the lower (eastern) part of the watershed.

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

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

Phosphorus is strongly adsorbed to sediment, which was reflected in Squaw Creek having both the greatest average sediment yields and greatest median concentrations of total P (0.14 mg/L) among the three watersheds. Squaw Creek's median P concentration is more than twice what EPA has proposed as the standard for Midwest streams. Concentrations of P in the South Fork, by comparison, had a median of 0.07 mg/L during

three years of weekly-biweekly sampling. Recent groundwater sampling from 24 wells located throughout the South Fork watershed has shown median and maximum total P concentrations of 0.030 and 0.340 mg/L, respectively. These groundwater P concentrations are found in similar materials and landscapes in Iowa (Burkart et al., 2004), and suggest that groundwater can also be a P contributor to streams.

Tailoring Conservation Practices to Watersheds and NPS Pollutants

Conservation practices used on row crop fields in the three watersheds reflect respective watershed characteristics and land use histories. A field-by-field assessment of conservation practices was conducted in each watershed to assess the variety and distribution of practices. An analysis of tillage practices, terraces, and contour farming shows the degree to which land managers have used these conservation practices to reduce nutrient and sediment losses from the three watersheds. Of the three tillage practices assessed (conventional tillage, mulch till, and no till),

mulch till was most widely utilized in all watersheds. Mulch tillage (>30% residue cover) was used on 62 to 91% of all row crop fields, whereas no till was used on 8 to 16% of the fields. Conventional tillage (<30% residue cover) was rarely used in Sny Magill and Squaw Creek, but was used on 30% of cropland in South Fork. Erosion losses from crop fields in South Fork are not a major concern in the flat, till plain portion of the watershed. In areas of the Northern United States, with relatively flat terrain and poorly drained soils, many producers still view conventional tillage as the most viable practice because the exposed soil is warmed faster in spring, often allowing earlier seeding and emergence of the crop.

In the Sny Magill watershed, contour farming, terraces, and other engineered structures are prevalent practices for reducing sediment losses from the steeper slopes in that watershed. Although row crop fields occupied only 26% of the land area in Sny Magill, most are terraced (77%) and/or farmed using contour planting (92%). Other engineered conservation practices are also used extensively throughout the Sny Magill

watershed, including a total of over 150 sediment basins and grade stabilization structures. Terraces are not as common in Squaw Creek (23%), but half the farm fields are planted on contours. Terraces and contour farming are not common in the South Fork watershed; fields with terraces occupy less than 10% of the watershed's cropland.

A Tale of Three Watersheds - revisited

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

drainage in the watershed make Sny Magill watershed perhaps the most vulnerable among the three streams evaluated.

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

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

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

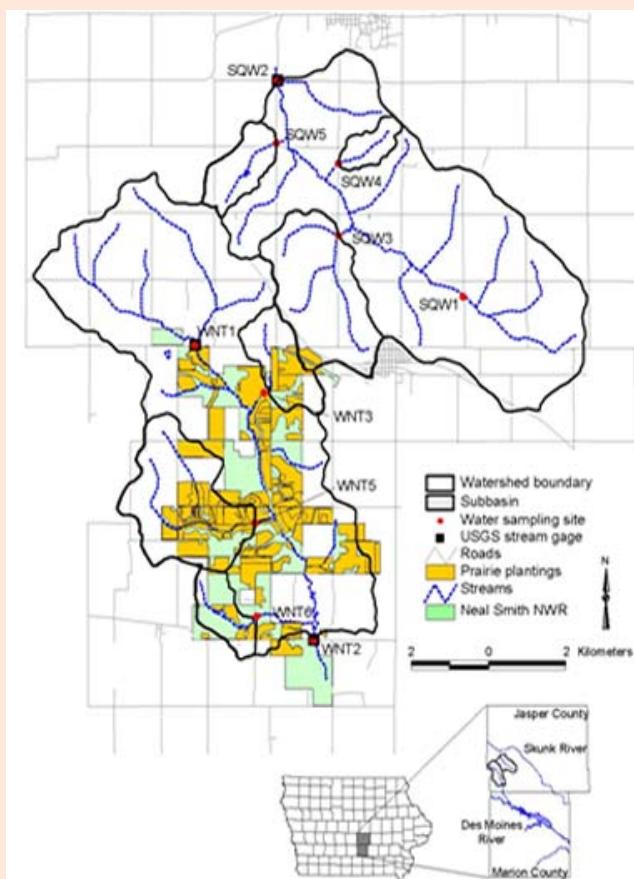


Figure 5. Extent of Prairie plantings in the Neal Smith National Wildlife Refuge within the Walnut Creek watershed.

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

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

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

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

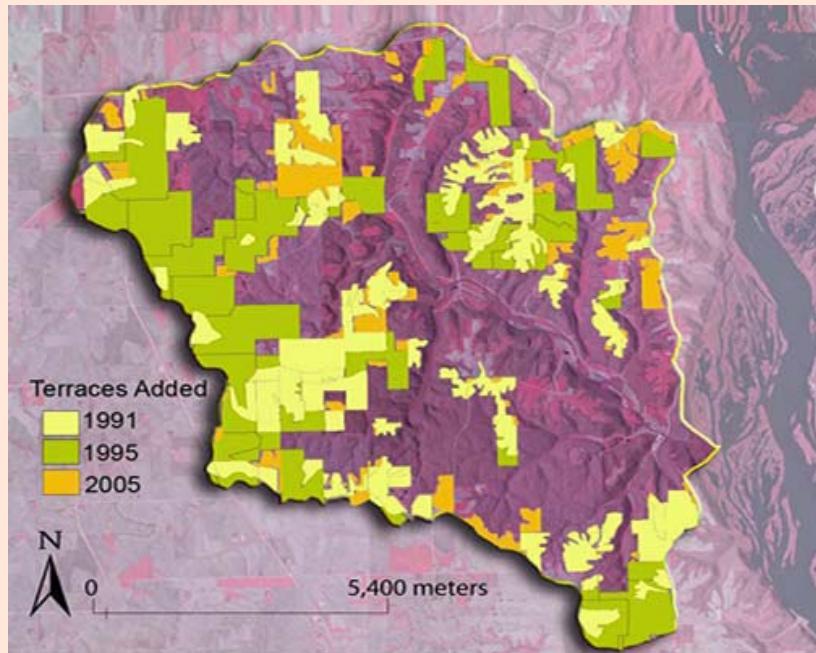


Figure 6. Cumulative additions of terraces to specific land tracts in the Sny Magill Creek Watershed.

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

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

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

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

A one-time inventory of conservation practices in the South Fork watershed was conducted during 2005. We compared the distribution of no-tillage management and CRP plantings with the distribution of HEL, which occupies 12% of the watershed (Figure 7). Very little (2.4%) of the watershed's cropland had been enrolled into CRP by producers, partly because this is some of the most productive rain-fed agricultural land in the U.S. While CRP has also been used to install buffers along streams and around livestock facilities, there has been apparent success in targeting of CRP towards HEL. That is, the proportion of HEL in the watershed in CRP is 4.6%, as opposed to only 2.2% of non-HEL (Table 2). The same is true of no-tillage practices that are highly effective in controlling erosion: although relatively few producers in this watershed have implemented no-tillage, largely due to concerns about planting delays during wet, cool spring conditions, a greater proportion of HEL (11.3%) is under no-tillage than is non-HEL (6.7%). There is little comparative data to evaluate

whether these practices are better targeted towards HEL in this watershed than in other areas. However, targeting success may also be indicated if conventional tillage practices that increase soil susceptibility to erosion have shifted away from HEL as these conservation practices were implemented. This does not appear to be the case, as conventional tillage occupies nearly the same proportion of HEL and non-HEL cropland, to within 2%.

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

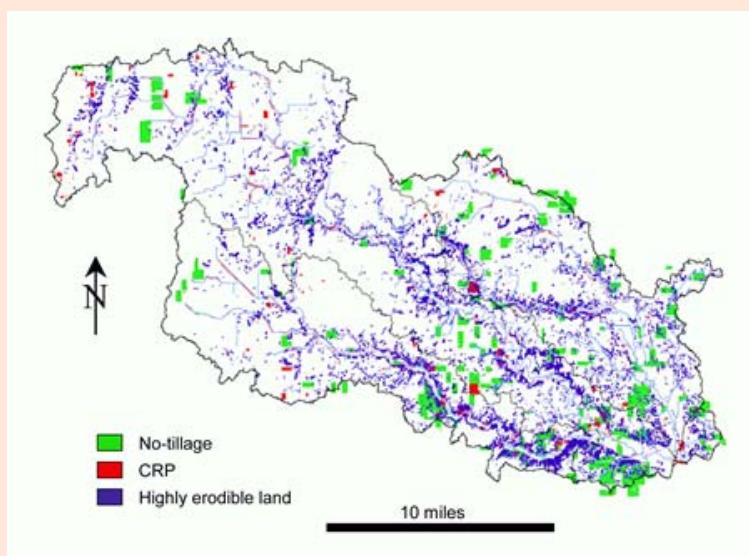


Figure 7. Distribution of key conservation practices for erosion control in the South Fork watershed, compared to the distribution of Highly Erodible Land.

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

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



Privatizing Ecosystem Services: Water Quality Effects from a Carbon Market

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

JEL Classification Code: Q25

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

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

The example we consider here is the case in which a carbon market that would allow U.S. farmers to receive payment for sequestering carbon when they retire land from production is implemented. This could occur if the

United States were to unilaterally implement such a market, or if at some future time the United States chose to sign on to a Kyoto-like accord, where carbon sinks were allowed to generate credits that could be traded to meet mandatory carbon reduction requirements. Under such a scenario, land retirement decisions would be driven by the prices paid for carbon and the amount of carbon that a particular parcel could sequester (we abstract from the important question of measuring the carbon storage potential of each parcel see the excellent work of Mooney et al., 2004).

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

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

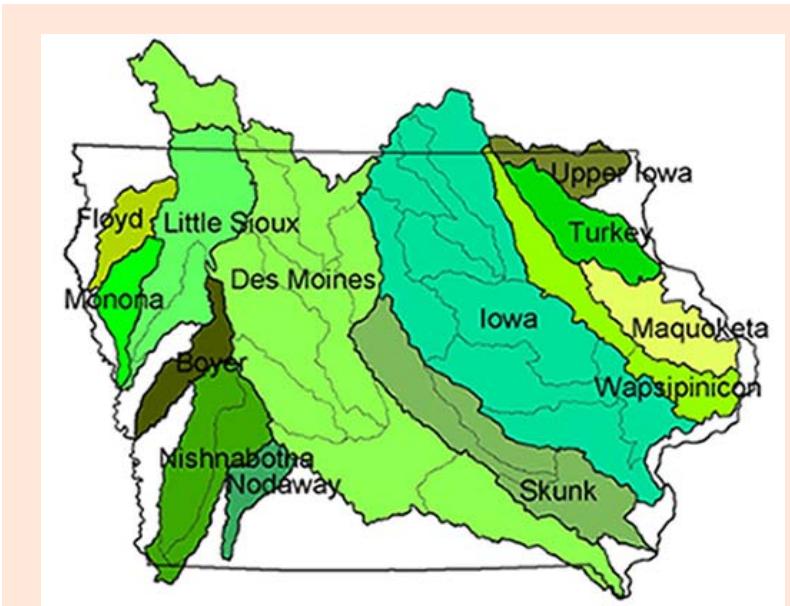


Figure 1. Study area and watershed delineations.

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

A Bit about Our Data and Models

To develop our models, we draw heavily from the National Resource Inventory to provide data on the land use, cropping history, and farming practices in the state of Iowa. This inventory is the most comprehensive data set on land use in the United States, and we use data on the 14,472 physical points in Iowa that represent cropland. Conceptually, our data and models are based on individual pro-

ducer and farm-level behavior, and we treat an NRI point as a producer with a farm size equal to the number of acres represented by the point (the expansion factor provided by the survey). Figure 1 illustrates the 35 watersheds corresponding to the United States Geological Survey 8-digit Hydrologic Cataloging Units that are largely contained in the state and are modeled in this study. To compute the amount of carbon sequestered when a land unit is retired from cropland, we rely on estimates from the Environmental Policy Integrated Climate Model version 3060. When land is retired from crop production, we assume that annual grasses are planted and maintained on the land, and we run a 30-year simulation to predict the carbon sequestration level associated with this change.

In addition, we also rely on estimates from a watershed-based model to assess the conservation policies. Unlike carbon sequestration, the degree to which land retirement improves in-stream water quality depends on critical interactions

between land uses in different locations within a watershed. Thus, for otherwise identical tracts of land, more water quality improvement may occur from retiring a piece of land from production that is located downstream from numerous other cropped points relative to one that is not. The potential filtering effect is just one example of the physical processes that need to be captured to assess the in-stream water quality effects of land retirement.

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

Water Quality Effects of a Carbon Market

To demonstrate the possible consequences of a carbon market that pays farmers for the sequestration of carbon in agricultural soils on water quality, we consider a simple scenario. Suppose that through an active carbon market, the price of carbon is such that about 10% of Iowa crop-

land is retired; suppose further that the cost of retiring all land within the state is about the same. While rental rates for farmland do vary across the state, they vary relatively little with respect to productivity (see Secchi & Babcock, forthcoming), and this simplifying assumption allows us to focus on environmental outcomes of the scenario without overly complicating the analysis. Under these assumptions, the cropland that will be removed from production will be the land that produces the highest carbon sequestration benefits per acre as this land will earn the highest return from carbon sequestration credits. In consequence, the land removed from production may or may not represent significant portions of the watersheds under consideration.

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

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

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

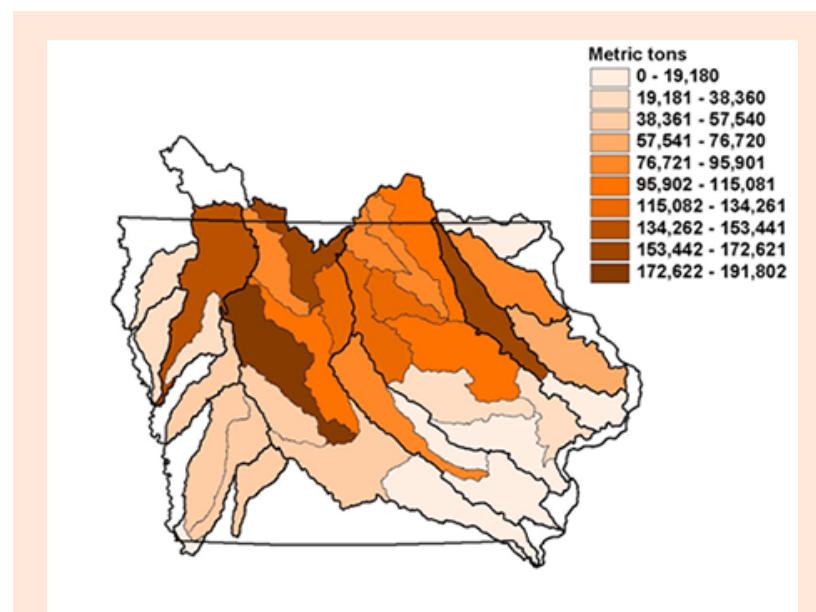


Figure 2. Carbon sequestration from carbon trading.

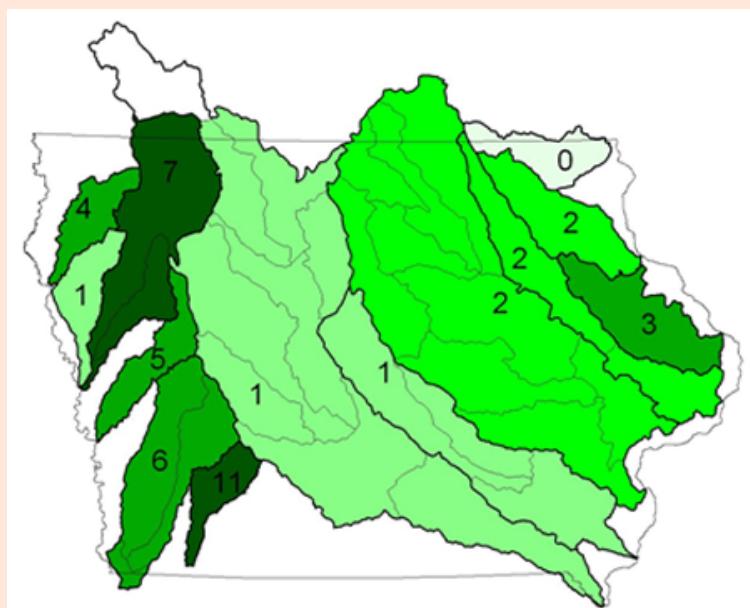


Figure 3. In-stream sediment changes from carbon trading (percentage reduction).

We find that for the two largest watersheds whose sediment dynamics is influenced by the presence of large reservoirs, the Des Moines and Iowa River watersheds, there is only a little improvement in sediment. In contrast, there are larger reductions in sediment in the South-Western part of the state, likely because the land

that is retired from production there is more erodible. In general, however, sediment reductions in percentage terms are lower than the reductions in nutrients, because land that has high carbon sequestration potential also has good productivity levels and is, therefore, more heavily fertilized.

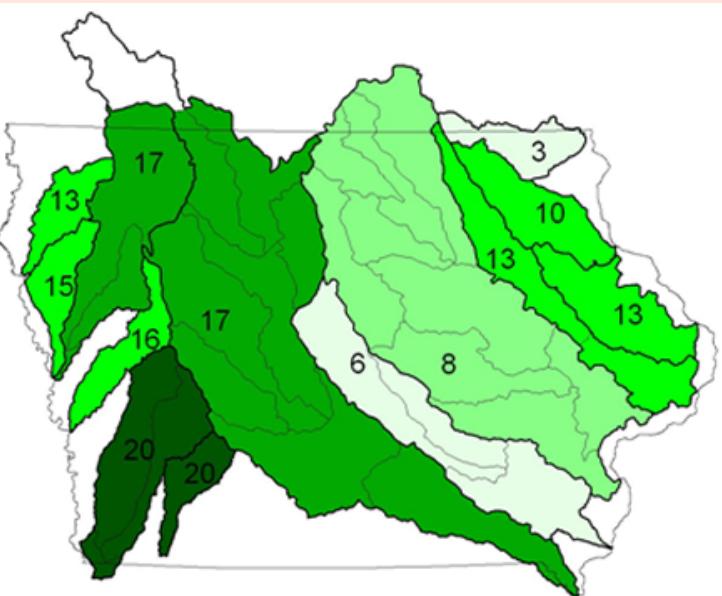


Figure 4. In-stream nitrogen changes from carbon trading (percentage reduction).

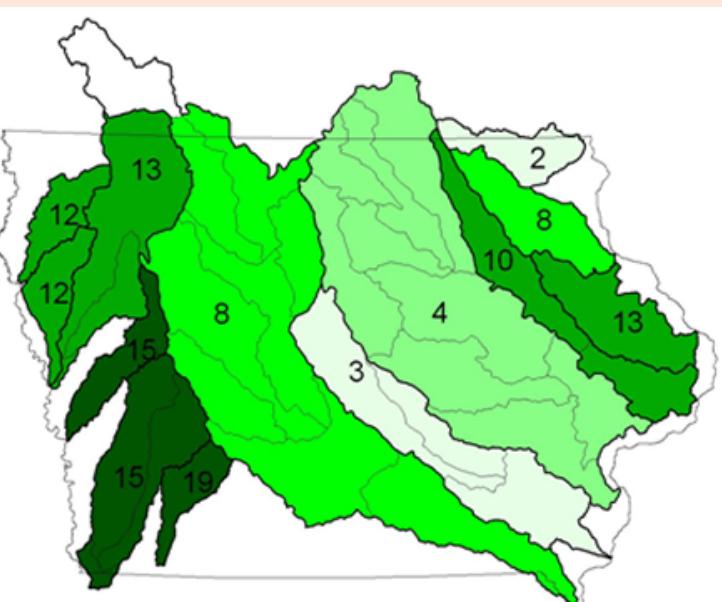


Figure 5. In-stream phosphorous changes from carbon trading (percentage reduction).

The retirement of land generally improves the total N level as seen in Figure 4. The reason for the improved N levels in many cases, as mentioned above, is that the land taken out of production is largely prime agricultural land: heavily fertilized and reliant on tiled drainage sys-

tems. Some of the highest reductions, a total of over 10,000 tons annual average, are in the Des Moines River watershed, which comprises large parts of the Des Moines Lobe and includes some of the most productive land in the state where most of the acres of land retirement are located.

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

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

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Nitrate Reduction Approaches

By Christopher Burkart and Manoj K. Jha

JEL Classification Code: Q25

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

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

Watershed Background

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

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

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

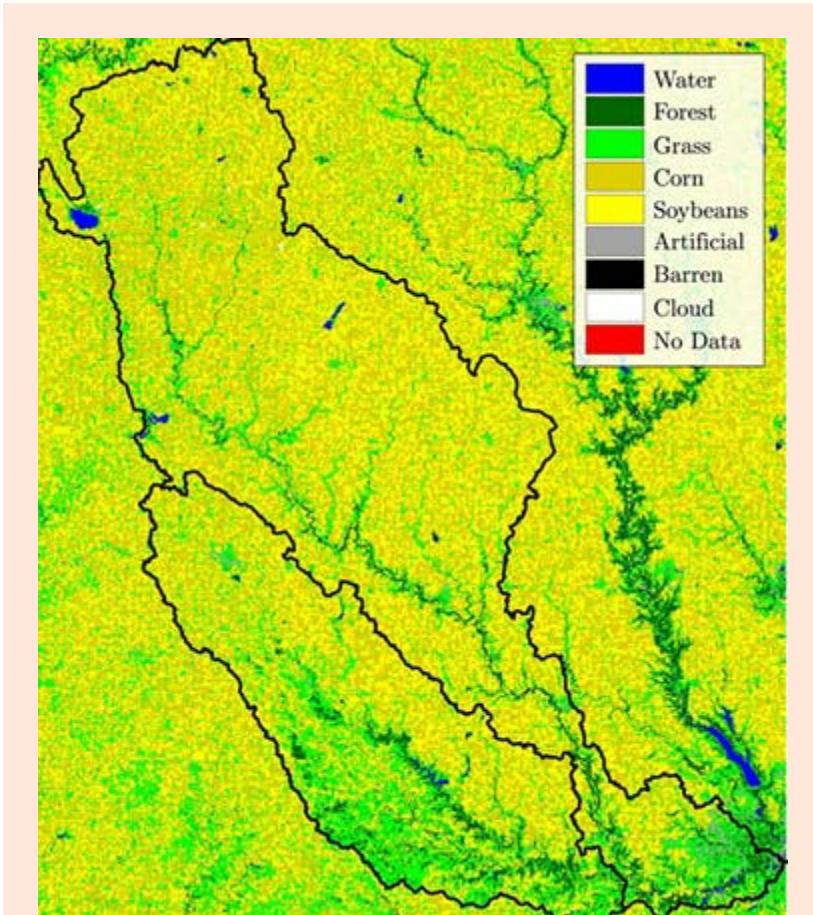


Figure 1. Land use in the Raccoon watershed.

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

Modeling Approach

While drinking water standards are given high importance due to their direct effects on human health, high nitrate levels cause other problems. However, control of ambient water

pollution in this watershed is still being developed, and there are no existing regulations outside of drinking water standards. Ameliorating problems such as hypoxia and nitrate toxicity for aquatic animals would require both a lower threshold for nitrates and complete removal of the nitrate from the watershed. Meeting the latter requirement with the technology currently used for drinking water purposes is inappropriate as it reintroduces the nitrate to the environment. The analysis here proceeds in the framework of existing regulations and the technology currently in place, but it is important to note that there are other impacts that merit consideration: namely, the effects of nitrate levels outside of drinking water considerations. Upland fertil-

izer reductions prevent nitrates from entering waterways in the first place, and have positive effects beyond contributing to drinking water standard compliance.

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

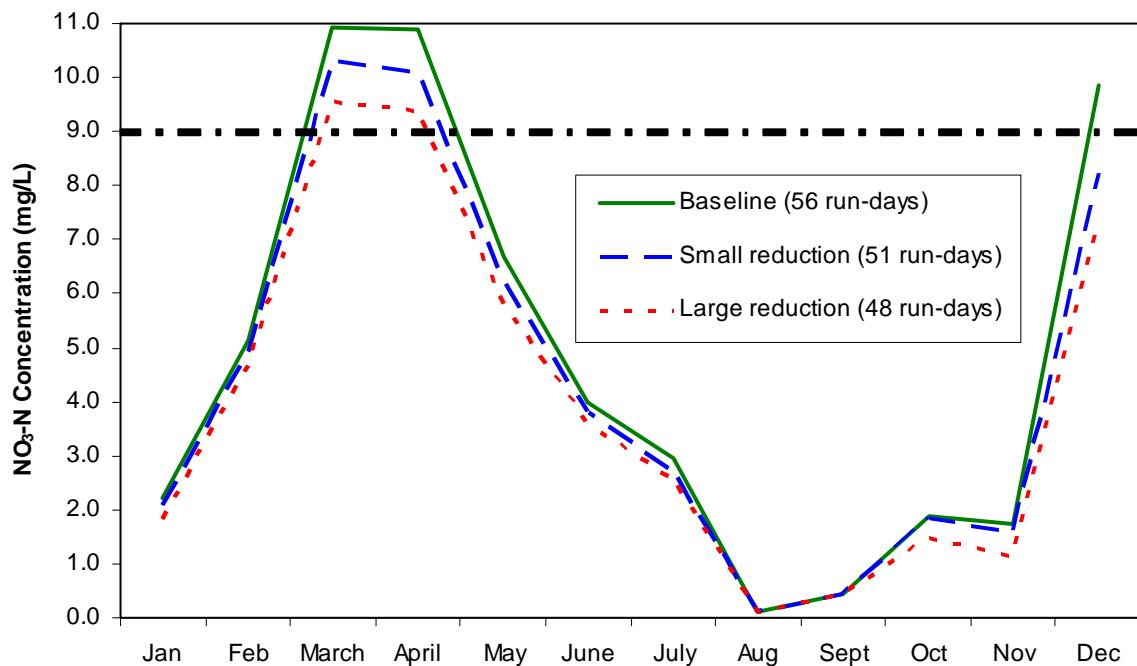


Figure 2. Nitrate loads by monthly average.

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

Policy Simulations

Three scenarios are run through the modeling system described above. One is a baseline in which the economic model leaves prices and nitrogen fertilizer applications unchanged, and the water quality model predicts the associated nitrate concentrations at the watershed outlet. The other two scenarios represent reductions in fertilizer applications simulated by the imposition of a nonpoint source trading scheme. This scheme works as follows: each farm is allocated fertilizer application permits for the total acreage it farms; for example, a 100-acre farm might receive 12,000 pounds worth of permits if the permit level is 120 pounds per acre. A farm has three choices in using its

permits. One is to apply exactly the permitted amount. Another is to apply less than permitted, and sell the surplus permits to the third group, those who purchase permits in order to apply at greater levels than initially permitted. Farmers make their choice of total application according to the model, taking into account the prices they face, their soil type, and the market price of a permit, which is determined by the distribution of farmer types. The total watershed application is reduced as long as the total permit allocation is smaller than the total amount originally applied. For purposes of simulations, this is done at two levels of permit allocations. From a baseline average application rate of 135 pounds per acre, one scenario restricts the per-acre permit allocation to approximately 120 pounds per acre and results in a simulated 6% reduction in annual load of nitrate at the watershed outlet. The

other restricts the allocation to approximately 108 pounds per acre and results in an approximate 12% reduction in annual nitrate load. These reductions are the result of the total mass of nitrogen being applied in the watershed being reduced.

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

To compare the water quality changes resulting from the imple-

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

Trading nitrogen permits between point and nonpoint sources can lower costs of reductions (Randall & Taylor, 2000). This is usually considered in the context of a non-point source generating excess permits by purchasing upland reductions. In that type of trading arrangement, a trading ratio is established to equilibrate a pound of upland reduction to a pound of point source discharge. Conceptually, this approach could work in reverse as well: nonpoint sources could generate permits for themselves by paying

for the removal system. While these trading opportunities are attractive possibilities, a quick look at the difference in costs in this case suggests that it would be much more efficient to simply run the nitrate removal facility a few extra days rather than implement any restrictions on farmer application. Five run-days at \$3,000 per day is \$15,000, far less than the \$160,000 in losses that would be incurred by farmers. Eight run-days of the nitrate removal facility are likewise much less expensive than the \$700,000 in losses associated with the stricter cap-and-trade policy.

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

There are also combinations of reduction strategies that could result in superior reductions with similar costs, even in the existing framework.

Coupling reductions with buffer strips, grassed waterways, changes in tillage, and application timing all can contribute to reductions in nutrient loads to a watershed. In addition, a more complete comparison would require information on possible deterioration of the nitrate removal media and the associated replacement costs, though these are at present uncertain.

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Organic Demand: A Profile of Consumers in the Fresh Produce Market

by John Stevens-Garmon, Chung L. Huang, and Biing-Hwan Lin

JEL Classification Code: Q13

Demand for organic produce in the United States has increased steadily since the early 1990s. In 2000, for the first time, conventional supermarkets sold more organic food than any other venue (Dimitri & Greene, 2002). According to the Organic Trade Association (OTA), organic food sales in the United States totaled \$13.8 billion in 2005, making up 2.5% of the retail food market. This is an increase from 1.9% in 2003 and from 0.8% in 1997 (OTA, 2006). This increase coincides with the implementation of national organic standards by the USDA in October of 2002, which provided uniform labeling for consumer recognition. Demand trends are expected to continue as more conventional retailers take up a larger portion of the organic market. Sales of organic foods are estimated to rise to \$23.8 billion by 2010 (NBJ, 2004).

The phenomenal growth in organic sales in recent years has brought additional farmland into organic agriculture industry. Dimitri and Greene (2002) estimated that between 1997 and 2001, U.S. farmers and ranchers nearly doubled the acreage of certified organic land, totaling to 2.3 million acres. With increasing production and supply of organic produce and meats, organic food, once considered a niche product, has become more available and affordable for consumers in mainstream grocery stores. It is estimated that 46% of total organic food sales are now handled by the mass-market channel, which includes supermarkets, grocery stores, mass merchandisers, and club stores (OTA, 2006). A popular perception tends to suggest that most organic consumers are white, wealthy, and have young children. However, the consumer base of organic food appears to have become more diverse and cannot be easily pigeonholed as the market is growing with

increased availability and popularity. A study by the Hartman Group (2002) found that half of the respondents who purchased organic food frequently have an annual income below \$50,000, and that African Americans, Asian Americans, and Hispanics purchase more organic products than Caucasians.

Our analysis used the Nielsen Homescan data from 2001 and 2004 (Box 1) to determine the characteristics of organic consumers, what they buy, how much they spend, and the price premiums they pay for organic produce. These two years give us a sample from before and after the implementation of the National Organic Program's (NOP) labeling standard. We focus on fresh produce because produce represents the largest sector, at about 39% of the organic market (OTA, 2006). One may speculate that the growing popularity of organic consumption could be attributed at least partially to the implementation of NOP. However, it is not our intention to contribute to the debate on the effect of NOP, mainly because Homescan data are not suitable for examining such an issue. We simply present a cursory look at the data to examine whether any notable changes have occurred after NOP by comparing household purchases of fresh produce in 2001 and 2004.

Who Buys Organic Produce?

Of all demographic characteristics, race seems to be the most correlated with organic expenditures. In 2001, we found that Asian Americans, compared to other ethnic groups, spent the most food dollars to purchase organic produce on a per capita basis. Though they bought comparable amounts of fresh produce, Asian Americans, on average, spent more on organic produce than White, Afri-

Box 1. The Sample Data and Description.

The Nielsen Homescan panel data include purchases of both random-weight and Uniform Product Code (UPC) food items. According to Nielsen, the panel consists of representative U.S. households that provide food purchase data for at-home consumption. For 2001 and 2004, more than 8,164 and 8,430 households, respectively, participated in the Homescan panel. In general, panelists report their purchases weekly by scanning either the UPC or a designated code for random-weight (unpackaged) products of all their purchases from grocery stores or other retail outlets. For packaged or UPC-coded food products, organic produce can be identified by the presence of the USDA organic seal or with organic-claim codes created by Nielsen. For random-weight items, the descriptions of designated codes can be used to identify organic produce. Homescan panelists do not report prices they pay for each food; they report total quantity and spending for each food. In addition, the Homescan data include product characteristics and promotion information, as well as detailed socio-demographic information of each household. For our analysis, household spending on selected fresh produce was calculated as average expenditures on per purchase record basis. Prices for organic and conventional produce were derived as unit values based on the household's reported expenditures and quantities. Average household expenditures on fresh produce were expressed in terms of per capita to control for household size effect. Furthermore, all expenditures and prices were expressed on nominal current dollars for 2001 and 2004, respectively. The Consumer Price Index for food and beverages increased by 7.49% from 2001 to 2004, and this inflation rate is used to calculate changes in real terms.

can, or Hispanic Americans. By 2004, Asian Americans' expenditures on organics fell, while White, African, and Hispanic Americans increased their spending on organic produce (Figure 1). Further, African Americans have replaced Asian Americans to become the ethnic group that spent the most on organic produce. The proportion of African Americans who purchased organic produce also increased from 34% in 2001 to 37% in 2004, while the proportion of organic users among other groups have remained relatively the same. These findings are in general agreement with the report that Asian, Hispanic, and African Americans are the ethnic groups more likely to purchase organic foods than Whites (Hartman Group, 2002). According to a more recent study by the Hartman Group (2006), Asians and Hispanics are motivated primarily by family concerns in buying organic products.

Organic expenditures vary by region. We found that in 2004, households in the Western United States purchased more organic produce than those residing in other regions, spending on average about \$4.90 per capita. This spending

amount represents an increase of 19% over 2001, after adjusting for inflation. Households residing in the northeastern and southern regions also registered an increase in average per capita spending on organic produce. The Central United States showed the lowest average per capita expenditure in 2004, which remained virtually unchanged from 2001. In terms of proportion of households that purchased fresh organic produce, the western region also showed the largest increase (almost 4%) of organic users from 2001 to 2004. The West and South appear to be the two fastest growing markets for organic produce in the United States.

According to Homescan data, the average per capita spending on organic produce increased by 12% in real terms between 2001 and 2004. As shown in Figure 2, this increase in spending is observed for all households across various income groups. It is interesting to note that average per capita spending on organic produce exhibited a U-shape relationship with income for households earning less than \$45,000 annually. Among households earning \$45,000 and more, organic spending appears

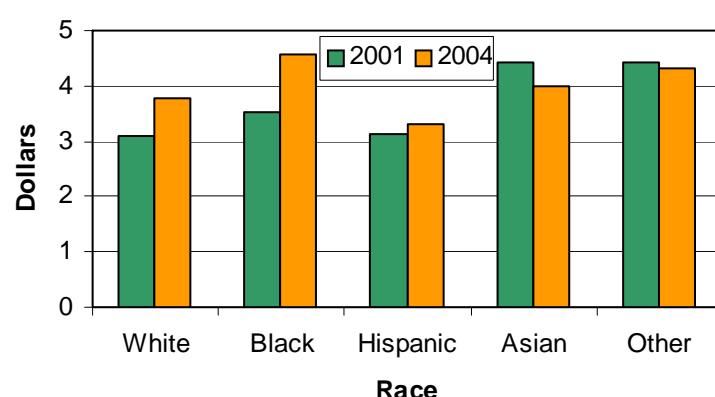


Figure 1. Average per capita spending on fresh organic produce for home consumption by race.

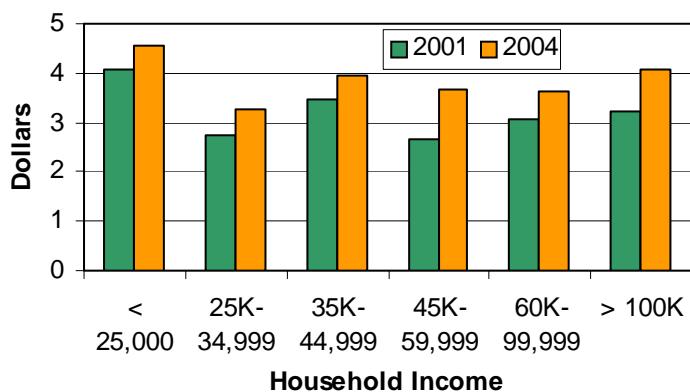


Figure 2. Average per capita spending on organic produce for home consumption by income class.

to rise with income. These patterns between household income and organic spending are observed for both 2001 and 2004. It is somewhat surprising to find that households with the lowest income level of less than \$25,000 spent the most—more than \$4 per capita on organic produce in 2001 and 2004. Furthermore, households in the \$35,000–\$44,999 income bracket spent about as much on organic produce per capita as those households earning over \$100,000 annually (\$3.94 versus \$4.09 in 2004). For households with annual income at \$25,000 or above, there appear little variations on average per capita spending on organic produce in 2001 and 2004. Overall, there is little consistent association between per capita expenditures on organic produce and household income. Studies suggest that lower income families choose to buy organic when possible as a means of preventative medicine, and thus are at least as likely to purchase organic as other income groups (Hartman Group, 2003; OTA, 2004).

The lack of a clear positive association between organic expenditure and income level may have prompted Laurie Demeritt, President of the

Hartman Group, to observe that “income is about the only thing that doesn’t skew at all by user and non-user. You get little skews in age, little skews in geography, little skews in education, but there’s nothing at all for income, so we don’t even look at that any more” (Fromartz, 2006). A recent survey conducted by the Food Marketing Institute (2004) showed that only 11% of organic shoppers polled bought organics at a natural-food supermarket, while 57% bought at mainstream grocery stores and discount stores. The fact that mainstream grocery stores are replacing the specialty food stores as the major outlets for organic foods could explain the seemingly fading relationship between organic expenditure and household income. It appears that income may no longer be a good predictor to profile organic consumers as the industry continues to grow and evolve into maturity.

What Do Organic Consumers Buy and How Much Organic Premium Do They Pay?

According to Homescan, tomatoes, potatoes, carrots, onions, lettuce, apples, oranges, bananas, grapes, and

strawberries were the top five vegetables and fruits in terms of their shares of fresh produce expenditures for home consumption. American households spent more on organic produce between 2001 and 2004 for all produce except oranges and lettuce. Overall, average per capita spending on these organic fruits and vegetables increased from \$1.64 in 2001 to \$1.91 in 2004, an increase of 8.5% in real terms. Tomatoes appear to be the most favored organic vegetable among American consumers with average per capita spending amounts 3–4 times those of other organic produce in both 2001 and 2004. Per capita spending on organic apples and lettuce held distant second and third places in 2001, while carrots and apples were ranked second and third, respectively, in 2004. Strawberries and bananas registered the largest increases in organic expenditures by 45% and 33%, respectively.

Since organic agricultural production is typically more cost intensive than conventional agriculture, many organic farmers rely on the premiums that organic foods carry to cover their extra costs. High premiums usually indicate high demand, signaling to producers which markets may be expanded. As indicated previously (Box 1), we calculated unit values (spending over quantity purchased) to derive price premiums for selected fresh produce because Homescan panelists do not report prices of organic and conventional produce. Thus, the organic premiums derived from unit values are not strictly the same as would be observed from the unit prices, if available. Except for oranges and onions, average organic premiums for the most valuable produce increased from 2001 to 2004 (Figure 3). In 2001, average organic premiums var-

ied from 1% (\$0.01/lb.) above the conventional produce for carrots to 78% (\$0.32/lb.) for potatoes. In comparison, organic premiums varied from 9% (\$0.08/lb.) for oranges to 78% (\$0.36/lb.) for potatoes in 2004. According to our calculations, organic potatoes carried a substantially higher price premium than other organic produce in both 2001 and 2004. This finding can be useful to organic producers who are looking for new crops to improve their profit margins. The changes in organic premiums between 2001 and 2004 were relatively moderate among the most valuable produce, except for lettuce and carrots.

In terms of dollar amount, average organic premiums that consumers paid in 2004 for apples, grapes, strawberries, tomatoes, and potatoes were fairly uniform at about \$0.35/lb. above their conventional counterparts. There are substantial variations among individual fresh produce, most notably in carrots and lettuce, which registered the largest increases in price premiums between 2001 and 2004. Tomatoes and apples also showed an increase in average price premiums by 52% and 75%, respectively. Overall, the average price premium for the selected produce increased from \$0.19/lb. in 2001 to \$0.29/lb. in 2004, which represents a 42% increase in real terms.

Price plays an important role in consumers' purchase decisions. A survey by Walnut Acres (2002) reported that 68% of consumers cited high prices as the main reason they did not buy organic foods. However, to many organic consumers, price could be of secondary consideration. They are willing to pay a price premium because they value and demand certain attributes from organic products. To them, the organic attributes are well worth the price difference. The

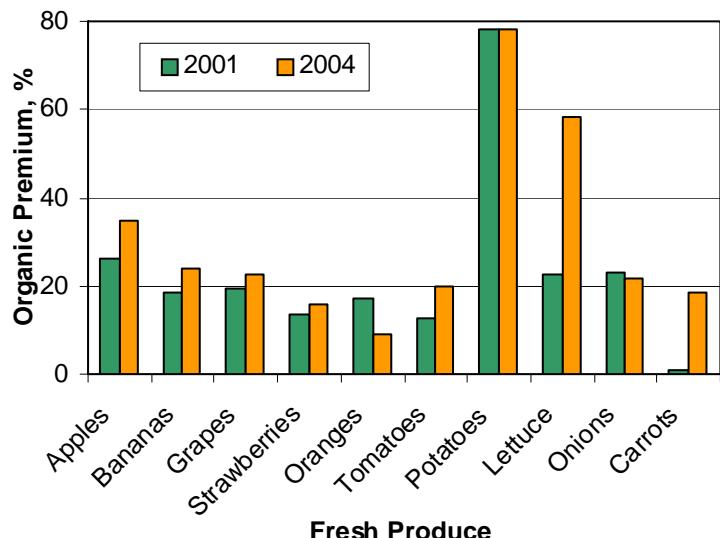


Figure 3. Average price premiums as % of conventional price for the top 10 most valuable organic produce.

fact that we find the organic premiums for most selected fresh produce increased from 2001 to 2004 suggests that the demand for organic produce remains strong, and consumers are willing to pay additional dollars for the organic attribute.

Based on limited data on organic prices over the period 2000–04 at the farmgate and wholesale levels, Oberholtzer, Dimitri, and Greene (2005) show that prices for organic varieties are comparatively more volatile than their conventional counterparts and organic price premiums were higher at the wholesale level than at the farmgate level. Of the three produce (broccoli, carrots, and mesclun mix) studied, they found that average annual organic price premiums at wholesale, as a percent of conventional prices, increased for carrots (143% to 148%) and for broccoli (141% to 153%) between 2001 and 2004, while the price premiums decreased for mesclun mix from 9% to 7%. It should be noted that the organic premiums calculated from the Homescan data are not directly

comparable with those reported in Oberholtzer, Dimitri, and Greene (2005). However, one would expect relatively lower organic price premiums at the retail level than at the wholesale or farmgate level as organic foods are becoming more competitive and increasingly marketed through mainstream supermarkets and discount club stores.

A Profile of Consumer by User Group

In our analysis, we categorized each household into user or nonuser group according to whether or not the household purchased organic produce. Then user households are classified into one of three user groups based on sample distribution of per capita spending on organic produce. In 2004, the first quartile of organic users with per capita spending greater than \$0 but less than \$0.75 is defined as light users, the second and third quartiles are defined as medium users (between \$0.76 and \$3.65), and the fourth quartile is the

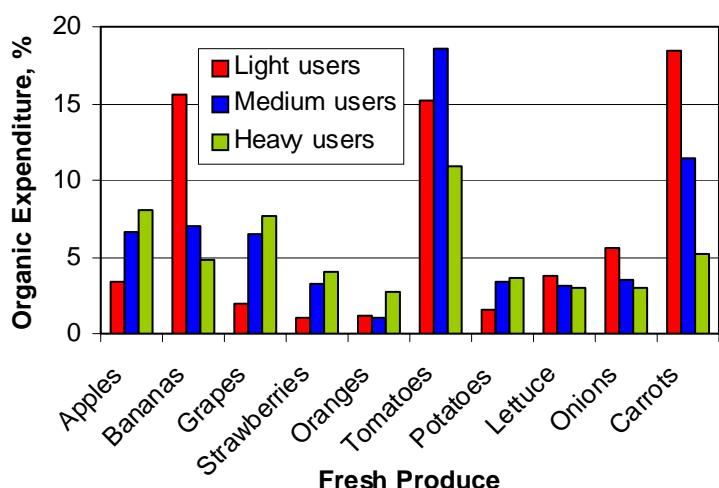


Figure 4. Market shares of selected organic produce by user group, 2004.

heavy users ($> \$3.65$). The nonusers account for 62.5% of the 2004 sample, while light, medium, and heavy users account for 9.6%, 18.6%, and 9.4%, respectively. In comparison, the proportion of user groups in 2001 are 62.9% (nonusers), 9.5% (light users), 18.4% (medium users), and 9.2% (heavy users). Overall, the result shows that proportionally more consumers have become organic users in 2004 than in 2001, with a slightly higher increase in both medium and heavy user groups. Per capita spending on organic produce by medium users increased from \$1.45 in 2001 to \$1.81 in 2004; an increase of 16% in real terms. For the light and heavy users, the growth in real per capita organic spending increased by 10% from 2001 to 2004.

With respect to market shares of selected organic produce across user groups, Figure 4 shows that light users spent the largest proportion of their organic expenditures on carrots, bananas, and tomatoes. The medium users purchased more tomatoes and carrots relative to other kinds of fresh produce, while heavy users seemed to

spend a larger proportion of their organic budgets on tomatoes, apples, and grapes. Overall, organic tomatoes appear to be the favorite fresh produce among the organic users, accounting for more than 15% of light and medium users' organic produce expenditure and more than 10% for heavy users. It is interesting to note that organic vegetables appear to be the preferred organic produce of light users, while the heavy users seem to have an affinity for organic fruits, especially apples and grapes. Heavy users buy proportionately more of each fruit than either the light or medium users, except for bananas. On the other hand, they tend to buy less of each vegetable than either the light or medium users, except for potatoes.

Comparing demographic information across user groups in 2004 gives us further insights in terms of how organic expenditures are related to these characteristics. As shown in Table 1, heavy and medium users have the largest proportions of those who have at least a bachelor's degree, while a larger portion of nonusers and light users have either a high

school diploma or some college. Interestingly, households whose heads have less than a high school education account for 1.9% of heavy users, the highest among all user groups. With respect to age, heavy users seem to comprise the largest proportion of the youngest households (household head age < 30 years), while the light users' group has the largest proportion of household head age between 30 and 49 years old. Medium and heavy users also have the largest proportion of older households relative to nonusers and light users, with the age of household head 50 years and older. Most heavy users are found in the Southern and Western United States, and the fewest are found in the central region. Medium users have the largest proportion of Whites relative to other user groups, while a relatively large proportion of Hispanic consumers belong to the light users' group. In comparison, heavy users are proportionally few among Whites, with the reverse being true for African, Asian, and other Americans.

Summary

We used the Nielsen Homescan data from 2001 and 2004 to analyze consumer purchase patterns of fresh organic produce. Our analysis shows that Asian and African Americans tend to purchase organic over conventional produce more than Whites and Hispanics. Households residing in the western region spent more on organic produce on a per capita basis than those residing in other regions. Contrary to popular opinion, we do not find any consistent positive association between household income and expenditures on organic produce. Although certified organic acreage has increased rapidly in

boosting the production of organic foods, our analysis suggests that demand appears to be growing faster than the supply so that organic price premiums for most selected fresh produce remained relatively high in 2004, varying from 9% for oranges to 78% for potatoes. Among all fresh produce studied, organic potatoes appear to command the highest percentage of price premiums in both 2001 and 2004.

We classified all households into four groups: nonusers, light users, medium users, and heavy users, according to their per capita expenditures on organic fresh produce. The proportion of consumers buying organic produce increased between 2001 and 2004, suggesting an increasing organic penetration. In terms of demographic characteristics, medium and heavy users are represented proportionately more by older households with the age of household head 50 years and older. Heavy users also comprise the largest proportion of the youngest households (household head age < 30 years), while light users have the largest proportion of household head age between 30 and 49 years old. In addition, we find that light users expend a relatively larger share of their organic expenditures on bananas and carrots than both the medium and heavy users. Organic vegetables appear to be the preferred organic produce of light users, while the heavy users seem to prefer organic fruits, especially apples and grapes. For all organic users, organic tomatoes are clearly the preferred choice over other vegetables.

Acknowledgements

The views expressed in this study are those of the authors, and do not nec-

Table 1. Selected Household Characteristics by User Group, 2004.

Category	Nonusers	Light Users	Medium Users	Heavy Users	Total
Educational Level (%)					
Less than high school					
diploma	1.86	0.86	1.66	1.90	1.73
High school diploma	19.60	15.80	15.02	11.53	17.63
Some college	31.49	31.60	30.73	26.36	30.88
College degree	32.05	35.56	33.61	32.83	32.75
Post-college degree	15.00	16.17	18.98	27.38	17.01
Age of Household Head (%)					
< 30 years	1.44	0.86	1.21	2.15	1.41
30–39 years	12.46	13.59	9.33	11.91	11.93
40–49 years	24.06	29.01	20.38	19.90	23.46
50–64 years	38.72	35.19	39.17	39.04	38.49
65 years and older	23.32	21.36	29.90	27.00	24.70
Region (%)					
Northeast	20.41	22.59	25.24	23.45	21.80
Central	18.76	16.79	14.25	10.65	16.98
South	41.36	37.41	34.89	31.43	38.85
West	19.46	23.21	25.62	34.47	22.37
Race (%)					
White	74.08	71.48	76.61	67.93	73.72
African	13.44	12.84	11.69	15.59	13.26
Hispanic	8.17	10.49	6.84	8.37	8.06
Asian	2.72	3.46	2.81	4.06	2.93
Other	1.60	1.72	2.04	4.06	1.92
Number of households	5,266	810	1,565	789	8,430

Note: User groups are classified based on sample distribution of average per capita spending on organic produce per purchase record. The first quartile is defined as light users with per capita spending greater than \$0 but less than \$0.75, the second and third quartile are defined as medium users (between \$0.75 and \$3.65), and the fourth quartile is the heavy users (> \$3.65).

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Water Quality Credit Trading and Agriculture: Recognizing the Challenges and Policy Issues Ahead

by Charles Abdalla, Tatiana Borisova, Doug Parker, and Kristen Saacke Blunk

JEL Classification Code: Q58

Economists have long championed market-based approaches over regulatory "command and control" approaches for addressing environmental problems. Recently, federal and state policymakers and some stakeholder groups have promoted market-based approaches for dealing with agricultural water pollution. At the federal level, the U.S. Environmental Protection Agency (US EPA) in 2003 issued a trading policy that allowed industrial and municipal point sources (PS) to meet their discharge requirements through purchase of "credits" from farmers and ranchers who implemented conservation measures that improved water quality. In October 2006, US EPA and the U.S. Department of Agriculture (USDA) reached an agreement to establish and promote water quality credit trading. In January 2007, USDA Secretary Johanns stated that in the upcoming farm bill, the administration will view market-based solutions as an important tool in federal environmental protection efforts aimed at agriculture (USDA, Natural Resources Conservation Service, 2007). At the state level, Idaho, Michigan, Ohio, Oregon, Pennsylvania, and Virginia have enacted laws or created regulatory programs to encourage water quality trading (see the summary of state efforts in ETN, 2007).

Given these activities, what can realistically be expected from market-based programs, and specifically water quality credit trading, in addressing the difficult issue of water pollution from agriculture? We conclude that policymakers are expecting far too much from trading as a tool to address agricultural nonpoint source (NPS) water pollution. Currently, water quality credit trading in agriculture is in its infancy and significant implementation

challenges exist. Trading program design and implementation must address complex physical, social, economic, legal and public policy issues. This will require more exchange among economists, policymakers, farmers, municipalities, and other stakeholders than is currently occurring. Only then will trading as a potential tool be fully understood and appropriately implemented.

In a previous issue of *Choices*, King (2005) examined issues encountered by water quality trading programs. We observe that changes in conditions affecting the supply and demand sides of potential water quality credit trading markets suggest the need to re-evaluate challenges that confront trading programs.

Non-Point Source Ag Pollution in the United States

According to the 2000 National Water Quality Inventory, agricultural NPS pollution is the leading source of impairment to rivers and lakes, and a major contributor to degradation of estuaries (Figure 1). Pollutants from agricultural croplands and livestock operations include excess fertilizer, herbicides and insecticides, sediment, and bacteria.

Controlling agricultural runoff is a longstanding and difficult problem. Agricultural NPS pollution loading is spread over large areas, and monitoring and measuring it is technically difficult and expensive. Agricultural runoff is highly variable due to the effects of weather variability, site-specific characteristics of the natural environment (e.g., soil type and land slope), and non-observable farm management practices (such as timing and precision of fertilizer application). While the cumulative effect of agricultural runoff can be observed through ambient water qual-

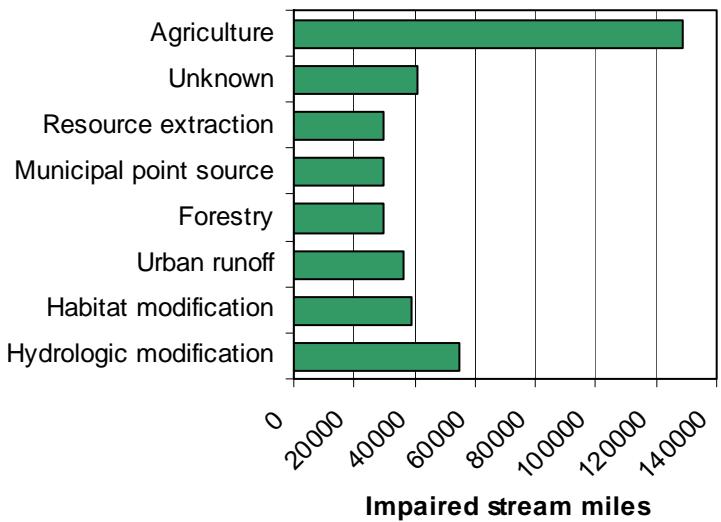


Figure 1. Leading sources of impairment of surveyed rivers and streams in the United States.

Source: USEPA, National Water Quality Inventory: 2000 Report, No. 841R02001, August 2002.

ity monitoring, it is generally impossible to trace the pollution back to specific farms. Existing computer models provide imperfect estimates of agricultural pollution loads. As a result, actual pollution amounts from a specific field or property are not fully known to regulators or farmers. Moreover, due to the variability in pollution loading, producers only partially control the runoff from their fields (Horan & Shortle, 2001; Braden & Segerson, 1993).

Accordingly, policymakers have long avoided environmental regulatory requirements for the agricultural sector. For example, the federal Clean Water Act excludes all agricultural sources (except for concentrated animal feeding operations - CAFOs) from federal regulation. Also, imposing environmental regulation will likely reduce agricultural producers' profits and may make U.S. agriculture less competitive than other nations with rules that are less stringent (Abdalla & Lawton, 2006). Instead of imposing environmental regulation, policymakers have offered incentive payment programs to

encourage farmers to voluntarily adopt environmental protection measures in the form of best management practices (BMPs) (NRCS, 2006).

This approach has failed to solve the water quality problems caused by agricultural runoff. Limited federal and state budgets constrain expansion of incentive payment programs for agricultural BMP implementation, and the existing programs have not always been cost-efficient (Babcock et al., 1995). At the same time, the policy of further reductions of point source (PS) pollution loads is no longer feasible. Increases in urban population bring about increases in pollution loading from municipal PS (wastewater treatment plants), and the necessary upgrades of industrial and municipal PS are costly.

Water quality credit trading policy seems to offer an easy solution to these problems. Economists have long argued that allowing PS to purchase pollution reduction credits from NPS will provide a low-cost alternative to PS upgrades (Baumol & Oates, 1988; Pearce & Turner, 1990; Faeth, 2000). Trading pro-

grams provide PS with flexibility in how to achieve their pollution loading limits, which creates incentives to discover cheaper and more efficient abatement methods. Credit sales could provide farmers with needed financial resources for BMP implementation. Trading is also attractive to policymakers and some stakeholders because it may provide private funds to supplement (or replace) federal and state incentive programs (King & Kuch, 2003).

However, agricultural pollution runoff does not meet the economic textbook definition of a tradable commodity. As a result, designing a water quality credit trading program poses a set of challenges. These are discussed in the next section.

Realizing the Potential: What Does Economic Theory Suggest as Critical Elements of a Water Quality Trading Program?

A water quality credit trading program is established to meet specific pollution reduction goals. Table 1 summarizes elements that are necessary for inclusion or consideration in the implementation of a program.

Critical elements of a water quality trading program. Even with these components in place, certain challenges must be addressed for a water quality credit trading program to operate. Many of the challenges relate to PS-NPS trades, where the regulated community (NPDES¹ permit holders) meets the unregulated community (agriculture and other NPS). In a 2005 issue of Choices, King examined the potential supply and demand for water quality credits. King states that, on

1. National Pollutant Discharge and Elimination System.

Table 1. Critical elements of a water quality trading program.

Public water quality goals	Set by federal, state, or local authorities based on public input and can be defined in terms of ecosystem function, fish population or public safety, or as surface water quality standards.
Pollution cap for a watershed	Limit on the total pollutant load from all sources to a water body. The justification for and size of the cap is based on the public's water quality goals. Usually the cap is set for an annual load of specific pollutants.
Regulated baseline for point sources or nonpoint sources	Numeric level of pollutant load allowed at a particular point in time. If all polluters meet their regulated baseline, the pollutant cap for the watershed will be obtained.
Unregulated baseline for agricultural nonpoint sources	Minimum level of pollution abatement that an unregulated agricultural operation must achieve before it can participate in a trading program. Sometimes called the threshold.
Credits	Units of goods (pollution reduction) to be traded in the water quality credit market. Credits are generated for every unit of pollution reduction beyond the baseline level.
Sellers (credit suppliers)	Dischargers that reduce pollution below the baseline and generate credits for sale in the market. Credits can also be sold by intermediaries, if allowed by program rules.
Buyers (demanders for credits)	Dischargers with regulated baselines for whom pollution reduction is expensive. For these sources, it is less costly to buy pollution credits from other parties and use these credits to help achieve their baseline loads. Credits can also be purchased by intermediaries and third parties, if allowed by program rules.
Trading ratio	Number of load reduction credits from one source that can be used to compensate excessive loads from another source.
Regulator	Entity that determines the water quality goals, establishes caps for pollutants in a watershed, approves and administers the trading program, and monitors and enforces the rules.

the demand side (PS), few dischargers are interested in buying water quality credits if the discharge restrictions are weak or un-enforced. More recently, many states have set limits for PS that are either already binding or are expected to become binding as populations grow. In some states, sources will be required to completely offset all new pollution loads. Thus, we expect that the demand for credits will change. In relation to NPS, King associates the lack of supply with agricultural producers' desires to avoid environmental regulation. King argues that by participating in trading programs, producers make the implicit admission that NPS pollution can be measured and controlled. As a result, some farmers are concerned that trading could lead to increased regulation. However, in

the past few years, a shift in perspective has been occurring. The federal and many state governments have passed regulations that require agricultural producers to implement practices to better manage runoff from their farms. Thus, producers are beginning to view trading as a way to hold off the implementation of future regulations. Despite these changes in conditions affecting both the supply and demand of potential water quality credit trading markets, other significant challenges still confront trading programs. Some of the key challenges are discussed below.

Challenges to Water Quality Credit Trading

Setting pollution caps. In order to ensure that a water quality credit

trading program achieves public water quality goals, a maximum loading or "cap" for each pollutant must be set for a watershed and enforced by the regulatory agency. While public water quality goals are often linked to services a water body provides (e.g. fish habitat), trading requires that a cap be defined for specific pollutants. This presents a challenge for accurately estimating the amount of pollution reduction necessary to achieve the public goals. In addition, many trading programs leave unregulated agricultural NPS out of the pollution cap, eliminating the link between public water quality goals and the program results (King & Kuch, 2003). Moreover, consistent enforcement of the cap is a necessary condition for trading.

Establishing allowable pollution limits (baselines). An unrestrictive cap on PS can diminish or eliminate the demand for credits. Conversely, setting a high baseline can reduce the NPS will or ability to produce an adequate supply of credits. Besides affecting the functionality of the credit market, assigning baselines raises the fairness issue since the parties with restrictive limits need to incur costs to achieve these limits. Baseline limits also raise questions about responsibility for pollution clean-up and about property rights of landowners. For example, many agricultural BMPs are funded with public cost-share money. A debate exists about whether BMPs installed with public funds are the property of farmers, and if so, whether these credits should be eligible for trades (Horan et al., 2004).

Theoretically, the agricultural baseline load should be linked to public water quality goals. This guarantees that the reductions beyond the baseline ("credits") reflect additional environmental benefits produced by

the source and supplied to the water quality credit market. In practice, the baseline is often set in relation to the current level of pollution, without regard to public water quality goals. In addition to jeopardizing public water quality goals, such baselines may create perverse incentives. For example, baselines may penalize those who have already implemented BMPs and reward those who have not by paying them for BMP implementation through credit sales (King & Kuch, 2003).

Complexities in establishing credits and associated risks with agricultural credits.

For NPS, pollution reduction from a BMP is difficult to accurately predict and monitor. The effectiveness of a BMP depends on its age, implementation factors, how well it has been maintained, and on site-specific conditions. Scientific models are often used to estimate load reductions from BMPs. However, imperfections persist in models and estimated reductions from a BMP likely differ from actual loadings. This complicates the process of credit verification and creates uncertainty about the magnitude of water quality improvements from a trade (Ribaudo et al., 1999). Also, requirements to improve credit verification processes and increase accuracy in pollution reduction estimation can significantly increase costs associated with credit trading. Consequently, the number of willing credit sellers and buyers may be reduced (King & Kuch, 2003).

In addition to these measurement and verification complexities, the uncertain nature of agricultural pollution reduction also implies that credit sellers (farmers) do not have complete control over the "goods" they sell (Shortle, 2007), while credit buyers "face the risk of having the

quantity bought falling below claimed level" (McCarl, 2006). In the majority of trading programs, variability in NPS pollution reduction is averaged and annual averages are used. The risks associated with agricultural credits are addressed in existing programs by requiring PS to purchase several NPS pollution reduction units to compensate for one unit of their own pollution increase (i.e., uncertainty trading ratio). However, such trading ratios implicitly increase the price the PS needs to pay for NPS pollution reductions. While the majority of trading programs employ ratios of greater than one, it has also been argued that trading ratios can be either less than or greater than one, depending on the variability of the agricultural discharges (Horan, 2001; Horan et al., 1999; Horan & Shortle, 2001).

Transaction costs. Transaction costs are costs that must be incurred to carry out a trade. Examples include the degree of difficulty in finding a buyer or seller, verifying credits, and negotiating and enforcing a trade. Trading will not occur if the transaction costs exceed the benefits of a potential trade (Stavins, 1995; Malik, 1992; Krutilla, 1999). Water quality credit trading programs that involve agricultural NPS are characterized by higher transaction costs than programs involving PS only. The transaction costs of finding a trading partner are higher because NPS are widely distributed across a watershed, and each source can generate only small numbers of credits in comparison with the larger demand of PS credit buyers (Woodward, 2006).

In addition to the costs of finding a trading partner, the measurement, verification, and enforcement of agricultural NPS pollution reduction can

be costly because of the nature of NPS pollution runoff (Woodward, 2006). Also, for all water quality credit trading programs, negotiating a trade can be difficult because of the novelty of the markets (Woodward, 2003). Unlike other environmental markets (e.g., wetland banking or SO₂ emissions trading programs), rules for water quality credit trading are not yet clearly defined and vary across programs. Many programs are complex, which increases the transaction costs for reaching agreements between potential credit trading partners. Examples of unclear and complicated rules include the credit certification process, credit resale, credit life span, monitoring and maintenance, liability, sale approval prices, and pricing. For example, in their survey of farmers and agency staff to assess perceptions of policies for NPS control, McCann and Easter (1999) found transaction costs for water quality credit trading to be problematic. The survey revealed that the trading program's administrative costs were perceived to be the fifth most expensive among the policies considered. Woodward (2003) suggested that transaction costs associated with NPS trades may decrease as program participants become more familiar with the rules and other trading partners, and become more confident in credit estimation and approval procedures.

Enforcing contracts and liability issues.

For the benefits of trading to be realized, there must be a mechanism to ensure that agreements arrived at are met. For example, in a PS-NPS trade, the potential buyers (PS) are liable for achieving pollution reductions as mandated by their NPDES permit limits. In contrast, the only document binding the potential sellers (NPS) is the private contract with the buyer. Most

existing trading programs hold the buyer responsible for monitoring the seller and enforcing the trade agreement. However, because the credit buyer and seller are more likely to focus on the credit price (as opposed to credit quality, i.e., delivering actual pollution load reductions), the regulator may bear more responsibility for verifying credits, and enforcing agreements (King & Kuch, 2003). By holding only the credit buyer liable for achieving pollution reductions, the regulator reduces the buyer's (PS) willingness to engage in an agreement. Suggested approaches to alleviate liability issues include the use of a mediator that can monitor and enforce the trading contract and place purchased credits in an "insurance pool" to guarantee that the NPDES limits are met even if one of the sellers fails to deliver the credits. The latter approach is used by the Pennsylvania and Ohio programs.

Leakage. Implementation of a trading program in one watershed or region can potentially lead to countervailing actions in areas outside the watershed. Stephenson et al. (2005) define leakage as an occurrence in which "a trade results in a net increase in loads." For example, it is proposed that some of the agricultural credits certified by Pennsylvania's trading program be generated by transporting manure/poultry litter to nutrient-deficient regions outside of the Chesapeake Bay watershed in Pennsylvania. If information concerning nutrient availability in soils in the receiving watershed is not well known, manure/litter importation can lead to increases in water pollution. Stephenson also provided an example about a farmer who installs a riparian buffer as a BMP, and generates and sells credits. However, to

The genesis of water quality credit trading in the Chesapeake Bay Region. Years of nutrient and sediment related pollution have caused significant impairments in the Chesapeake Bay. In 2000, the renewed Chesapeake Bay Agreement established ambitious targets for member states to significantly reduce their portion of nutrient pollution by 2010. These regional goals have had a ripple effect on each state in the watershed.

By 2003, Maryland instituted tributary strategies that placed caps on nutrients entering the state's waters. These caps were the impetus for Pennsylvania to seek new means for reducing its nutrient loads that flow across state lines. In 2006, Pennsylvania adopted its nutrient trading policy that enables PS-NPS trades. Pennsylvania is also attempting to ratchet down municipal sewage treatment plants' nutrient limits.

Virginia recently established a PS-PS trading program with the intent to eventually include NPS in trades.

Maryland and West Virginia are also in varying stages of developing state trading programs.

It is much too soon to judge how successful water quality credit trading will be in meeting the collective reductions necessary for improving and restoring the Chesapeake Bay.



compensate for reductions in productive land due to buffer implementation, the farmer expands the productive acreage in a different place, increasing the nutrient and sediment loads in that vicinity.

Scale of the trading program. Many existing water quality credit trading programs have been developed for relatively small watersheds. However, in a larger watershed, more opportunities exist to find a trading partner with significantly different pollution abatement costs. Thus, greater reductions in costs of meeting public water quality goals can be realized. Also, in a large watershed, a large number of buyers and sellers can help ensure that the market participants do not exploit the market power or distort efficient trading (Woodward, 2003; Hahn, 1989; King & Kuch, 2003). Currently, the regulatory driver for developing trading programs for larger regions is often lacking. Water quality standards or Total Maximum Daily

Loads (TMDLs), which are considered a driving force for trading programs, are usually set for small watersheds. The Chesapeake Bay Region is an exception.

Sizing up the Evidence

A variety of trading and other market-like programs have been created over the past 20 years. Failure to address the challenges identified above is a reason why many of these programs have been short-lived or have not resulted in much trading activity (Breetz et al., 2004). There are also examples of successful water quality credit trading programs. The Long Island Sound (CT) and Tar-Pamlico Basin (NC) trading programs both experienced relatively long lives and resulted in documented pollution load and cost reductions. The programs resulted in reallocations of pollution caps among PS and did not address the challenges posed by NPS runoff.

Alternatively, trading programs in the Miami Watershed (Ohio), South Nation River Basin (Ontario, Canada), Cherry Creek (Colorado), Beet Sugar Cooperative and Rahr Malting Pollutant Offsets (Minnesota), and Red Cedar River (Wisconsin) all involve both PS and agricultural NPS (Breetz et al., 2004). Some of the challenges associated with agricultural runoff have been addressed in these programs by creating an intermediary between credit sellers (NPS) and credit buyers (PS). Such intermediaries (referred to as aggregators, credit banks, or brokers) can reduce the transaction costs of finding trading partners, and credit verification and monitoring. Intermediaries may also potentially bear some liability for delivering pollution reductions specified in trading agreements. In existing programs, such intermediaries are a joint venture of the regulatory authorities and public and private entities. The funds used to purchase credits from agricultural NPS have been drawn from both PS and federal- and state programs. In other words, the programs are essentially hybrids between market-based trading programs and government-managed tax-and-subsidy schemes. The rules for selecting NPS projects to generate credits and for selecting prices that PS must pay differ among the programs, making some of them more like market-based trading and others more like government-directed offsets (Stephenson et al., 2005).

Conclusions

Unresolved Public Policy Questions

Water quality credit trading has been perceived by many as an alternative to command and control regulations. Yet, water quality regulation is a necessary driver for trades to occur.

Thus, trading alone cannot solve the challenges posed by largely uncontrolled agricultural pollution. A number of important public policy questions have been raised as discussions of trading have occurred. For the most part these questions remain unanswered or various interest groups and governmental agencies have answered them differently. Among the questions are:

Do the political will and resources exist? Do federal and state decision-makers have the political will and resources to enforce regulatory caps on PS and NPS? This is a critical step because the value of a water quality credit is dependent on the enforcement of this cap. Without an enforced cap, there is nothing of value for potential market participants to trade (King & Kuch, 2003).

Will government define the right to pollute and the right to clean water?

This question focuses on a key underlying issue in trading program design. In terms of trading, "you can't sell what you don't own". The answer to this question determines who pays and who benefits from trading. As noted, an example of the unresolved nature of this question is the debate over farmers' ownership rights to publicly funded BMPs. Opinions vary over assignment of property rights to private parties from publicly funded projects, raising questions about the water quality benefits from such "double-dipping." Allocating and enforcing property rights is a fundamental role of government. Are governments willing to reconcile property rights questions of this nature?

Will trading programs be accepted as equitable? Is it fair when one category of polluters – PS – have regulatory effluent limits placed on them while NPS are required to meet only

program-specific baseline requirements? King and Kuch (2003) suggest that PS dischargers believe that there is an inequitable allocation of pollution rights to NPS dischargers.

Will market approaches for environmental goods or services be accepted?

Some stakeholder groups oppose trading because they believe that it is inappropriate to put a price on natural and environmental resources and trade them in a market (Goodin, 1994; Hahn, 1989; Hahn & Hester, 1989; King & Kuch, 2003; Woodward, 2003). Some are suspect of market-based approaches and perceive trading as excluding environmental interests. Also, the terminology associated with water quality credit trading is not well understood by environmental, farming, and development communities. This may prevent them from effective participation in trading program development.

Where to from Here?

Recently, federal and state policymakers as well as some stakeholder groups have promoted market-based approaches to address the agricultural water pollution. We described some of these policy activities and raise the question: what can be expected from market-based programs, and specifically trading, in addressing the long-standing and difficult issue of water pollution from agriculture?

After assessing these challenges, we conclude that federal and state policymakers are expecting too much from trading as a tool to address NPS water pollution from agriculture. Since King's 2005, *Choices* article, institutions have made progress creating a more supportive trading environment. Nevertheless, the physical and regulatory context for agricultural NPS pollution does not match

the conditions that economic theory suggests are needed for widespread trading to occur. The majority of agricultural sources do not face an enforceable cap. Thus, it is difficult to ascertain whether PS-NPS trades will create new water quality improvements. King and Kuch (2003) state that PS-NPS trades cannot achieve water quality standards in watersheds where the NPS dischargers are responsible for the bulk of nutrient discharges or where very large reductions in nutrient loading are necessary. The lack of documented success in water quality credit trading adds credence to the idea that there is a mismatch of theory and practice.

We believe that water quality trading in agriculture should continue to be explored and be field-verified for its use as a tool to reduce the costs associated with pollution reductions. However, in the interim, policymakers must reduce their expectations and reliance on market-based solutions until there is more evidence that validates that these programs can help meet pollution reduction goals. Policymakers must recognize that water quality credit trading in agriculture is still in its infancy and that the challenges identified are not yet well understood.

Ongoing trading efforts should be regarded as experiments. Increased attention must be paid to designing future experiments to better understand the physical, social, economic, legal, and policy considerations. This will require greater exchange among economists, physical scientists, policymakers, farmers, municipalities, community members, and other stakeholders. As a more thorough knowledge of trading as a tool to address agricultural water quality problems is gained, its potential for

use to help meet public water quality goals increases.

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Farm Growth, Consolidation, and Diversification: Washington Dairy Industry

By Tristan D. Skolrud, Erik O'Donoghue, C. Richard Shumway, and Almuhanad Melhim

JEL Classification Code: Q12

The shrinking number of farms in the United States is well-documented. Between 1974 and 2002, the total number of farms in the United States declined by 21%. While this represented a large drop in the overall number of farms, the number of farms with milk cows declined much more dramatically, falling by 79% during this period (USDA/NASS, 2002). With four times as many milk cows per farm in 2002 than in 1974, it is obvious that the dairy industry has become much more concentrated. Further, the entire decline in number of farms with milk cows occurred in size categories with fewer than 500 cows. The number of farms with 500-999 milk cows grew by 36% and the number with 1,000 or more milk cows more than doubled. Changes in the State of Washington generally followed those of the Nation.

The growth in the number of the largest-sized farms creates the most intrigue for economists and policymakers alike. As one of the last bastions of nearly perfectly competitive production, does this growth in farm size hint at a major change in the historically competitive nature of agricultural commodity supplies? For example, in 2002, more than 30% of milk sales came from just 1.5% of dairy farms. This situation warrants careful attention since adverse environmental effects often accompany increases in farm sizes, particularly for confined animal operations.

While we know that significant changes are occurring in farm size, no one has yet identified which farms are growing or shrinking in size. Nor has anyone documented the extent of commodity diversification on farms of different size. Which farms grow? Do farms in the larger size categories actually grow the most rapidly? Or do medium-sized farms combine with other farms of comparable size to create new large organizations? Do farms in different

size categories increase or decrease their levels of diversification over time?

To answer these questions, we examined longitudinal data from the Census of Agriculture in 1992, 1997, and 2002 for dairy farms in Washington. This is an important industry in both the state and Nation. In the United States, dairy products rank second among all agricultural commodities in value of production (USDA/NASS, 2006a). Washington ranks 10th in the nation in milk production and first in milk production per cow, while the value of milk production in the state also ranks it second in importance among all agricultural commodities (USDA/NASS, 2006b). The state's dairy industry is highly concentrated, but geographically divided. More than half the milk cows are located in two counties; Whatcom on the west of the Cascades and Yakima on the east. The demographics are changing with rapid movement of cows to the east side of the Cascades. Cow numbers in Yakima County grew by more than 30% between 1997 and 2002, while those in Whatcom County declined.

Sample Selection and Information Collected

For our analysis, we included all farms for which the owner checked farming as his/her main occupation and for which at least 50% of all agricultural income (not including government payments) came from the sale of milk and dairy products. As a result, 781 farms are included in our sample, representing 65% of all Washington dairies in the 1992 census.¹ We ranked the farms from lowest to highest in terms of agricultural sales and then divided them into 10 equally sized cohorts. In other words, each cohort had the same number of farms in 1992 with the smallest 10% of dairy farms in the state in

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Table 1. 1992 agricultural sales.^a

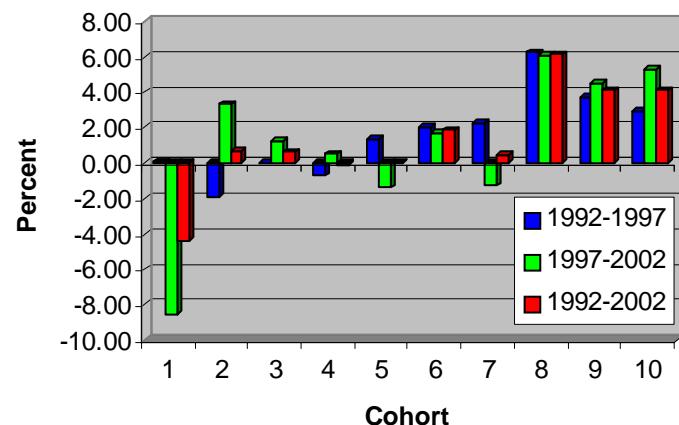
Cohort	Range		
1	<	95,000	
2	95,000	-	155,000
3	155,000	-	215,000
4	215,000	-	270,000
5	270,000	-	325,000
6	325,000	-	405,000
7	405,000	-	505,000
8	505,000	-	685,000
9	685,000	-	1,085,000
10	>	1,085,000	

^a Because of data confidentiality conditions, these ranges are only approximate.

Cohort 1 and the largest 10% in Cohort 10. The approximate range of sales for each cohort is reported in Table 1. Where possible, we tracked individual farms in each cohort over the next two censuses. We also created new cohorts for entrants in 1997 and 2002, for a total of 12 cohorts.

We recorded each farm's tenure status, total agricultural sales (exclusive of government payments), and milk and dairy product sales in each census year that it appeared. Based on this information, we calculated the number of farms in production, the number that entered and exited, farm size distributional statistics (mean, median, standard deviation, skewness, kurtosis, and range of sales), and the percent of cohort farms in each of four diversification categories. The percent of total farm sales (exclusive of government payments) derived from milk and dairy product sales determined the diversification categories: (1) 90% or more, (2) 75 - 89.9%, (3) 50 - 74.9%, and (4) less than 50%.

1. The remaining 35% of dairy farms consisted of retired and residential/lifestyle farmers.

**Figure 1.** Annual growth rates.

Farm Growth

Mean growth rates of 1992 dairy farms that remained in production varied considerably both among cohorts and between censuses. After adjusting for inflation between the censuses, the dairy farms grew at an average compound rate of 1.6% per year between the 1992 and 1997 censuses and 1.1% per year between the 1997 and 2002 censuses, averaging 1.4% between 1992 and 2002.

Figure 1 shows the annual growth rates we computed for each cohort for the 5- and 10-year periods. The average size of the smallest cohort of dairy farms decreased over the 10-year period, while the average size of farms in the three largest cohorts increased substantially and steadily over time. Farms in the intermediate size ranges generally grew slowly and more erratically. Overall trends suggest that, as farm size increased, so did the corresponding growth rate.

Distribution of Farms within Cohorts

Farms were close to being uniformly distributed within most cohorts in 1992. Only in the largest cohort was the distribution of farms appreciably

skewed. In this cohort, the majority of farms lay in the lower part of the range and only a small number of much larger farms resided in the upper end of the range. In successive censuses, as farms tended to grow in size, the surviving farms in all cohorts became positively skewed, similar to the largest cohort in 1992. This finding implies that a small number of farms in every cohort grew much more rapidly than others.

These results suggest that average cohort sales were particularly influenced by a small number of farms that grew rapidly within each cohort. In fact, in each of the five smallest cohorts, a majority of the surviving farms were smaller in each successive census than in 1992. Therefore, if used improperly, average farm size can result in very misleading conclusions.

Farm Size and Diversification

Because of the criteria used to select farms to include in the sample, no dairies in 1992 were in the most diversified sales class (with less than 50% of agricultural sales from milk and dairy products). As apparent from Figure 2, the smallest three

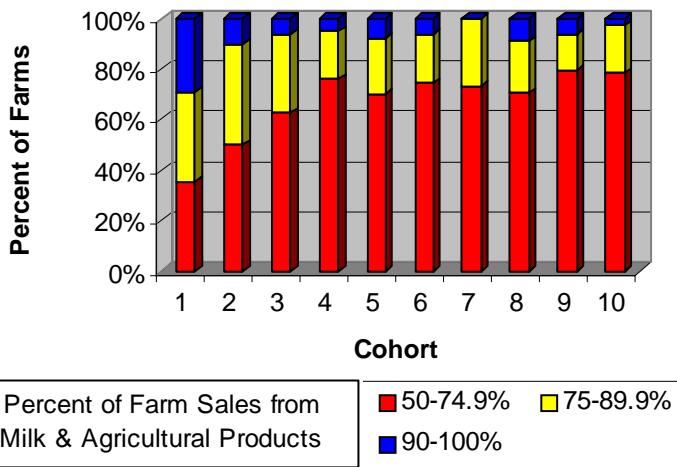


Figure 2. Farm diversification, 1992.

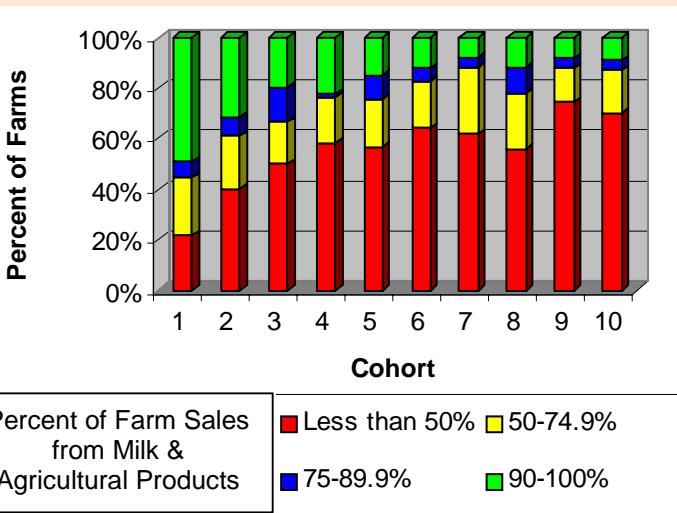


Figure 3. Farm diversification, 1997.

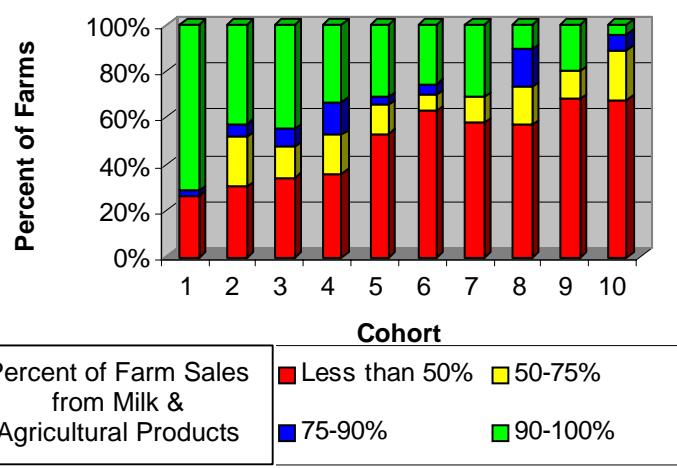


Figure 4. Farm diversification, 2002.

cohorts were the most diversified and all larger cohorts were more specialized.

In successive censuses, every cohort became more diversified. For example, the percent of farms that received 90% or more of their agricultural sales from milk and dairy products declined from 35% in 1992 to 27% in 2002 in Cohort 1 and from 78% in 1992 to 67% in 2002 in Cohort 10. Much more dramatic was the shift of farms to the most diversified sales class. By 2002, nearly 75% of farms in Cohort 1 received less than half of their agricultural sales from milk and dairy products, while none did in 1992.

Across cohorts, diversification followed roughly the same pattern in 1997 and 2002 as in 1992. The smallest cohorts were the most diversified and specialization increased with farm size (see Figures 2-4). We tested this graphical evidence by examining the correlation between farm size and level of diversification. Confirming our results, we found statistical evidence that as farm size increased, farms tended toward greater specialization. This tendency became stronger over time.

While the diversification trends between 1997 and 2002 followed those between 1992 and 1997, some caution should be exercised when interpreting the most recent statistics. Milk and dairy product sales do not include cull dairy cow or other cattle sales, and milk prices were lower in 2002 than in 1992 or 1997. Consequently, it is possible that part of the apparent increase in diversification in 2002 was due to a higher than normal culling rate induced by the lower milk price.

A further caution should be made about the diversification levels. We measure farm size by value of agricultural sales (exclusive of government

payments), and our sample was selected to include only those farms for which milk and dairy product sales accounted for at least 50% of agricultural sales. Consequently, the most diversified farms with milk cows did not enter our initial sample. If they had been included, the evidence of diversification within the dairy industry would be even greater.

Farm Entry and Exit

Between each pair of censuses, more than twice as many dairy farms exited the industry in Washington as new farms entered. Smaller dairy farms tended to exit at higher rates than did larger farms. In Cohorts 1-7, an average of 3.5 farms exited for each farm that entered between 1992 and 2002. In contrast, an average of just over one farm exited for every farm that entered in Cohorts 8-10, implying a very low net exit rate. Further, the largest farms (Cohort 10) had fewer exits than entrants, which resulted in positive growth in the number of largest dairy farms.

Farms of all sizes entered the marketplace. However, their distribution and behavior differed widely from incumbent farms. While their mean size was much larger than the incumbents, falling between the means of the two largest incumbent cohorts, their growth rates tended to be much slower than the growth rates of the largest incumbents. They averaged less than 1% growth per year. They also entered the marketplace with a higher average level of diversification than any of the large incumbent farm cohorts in the initial sample and continued to diversify at a much more rapid rate.

What Does All This Information Mean?

This analysis of longitudinal agricultural census data for the Washington dairy industry has produced important insights about the relationship between initial farm size and both subsequent growth rates and the tendency to diversify. The largest group of cohorts is growing the fastest, suggesting that, despite earlier evidence that economies of scale were largely exhausted by 750-cow farms (e.g., Matulich, 1978), dairy farms in the state are not yet converging toward a size that minimizes average cost within the current size range. However, the fact that it was Cohort 8 rather than Cohort 10 that grew at the fastest rate does suggest that economies of scale may be diminishing for the very largest farms.

Additionally, we found that the larger the farm, the greater the tendency to specialize. In other words, larger dairy farms derived more of their revenues from milk and dairy product sales, while smaller farms turned to a more diverse range of outputs to generate their agricultural revenue streams. The only exceptions applied to new entrants. While their average size was very large at entry, they were much more diversified than large incumbent farms and grew much more slowly. However, the average level of diversification in all cohorts has increased over the 10-year period examined. This finding is particularly surprising for an agricultural commodity that has been one of the last bastions of the single-product farm.

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Fruit and Vegetables Go Back to School

by John L. Park, Benjamin L. Campbell, Andres Silva, and Rodolfo M. Nayga, Jr.

JEL Classification Code: I38, Q18

Perhaps one of the most alarming trends plaguing our modern food system is the seemingly rampant increase in the prevalence of obesity across the United States. The Department of Health and Human Services reports that one in three adults is obese, and two out of three are considered overweight or obese. Even more alarming is the trend among children, where obesity rates have nearly tripled since 1980 (NCHS-CDC, 2006). Policy makers across the country have responded with efforts to drive foods of minimal nutritional value out of our schools and replace them with whole grains and fresh fruit and vegetables (Schmid, 2007; Zhang, 2007).

The resulting policies and programs may represent opportunities for marketers and producers of fresh fruit and vegetables to reach a growing market segment within our schools. However, it is not enough to simply provide an appealing product to students. Instead, successful marketers will appeal to the needs, perceptions, and preferences of those responsible for wholesale purchasing (Park, 2001). They need insight into the mentality of the school foodservice director. The effectiveness of these programs to improve dietary quality and presumably health is currently being debated. Externalities such as the influence of school foodservice buying habits and constraints may impact the effectiveness of these programs to achieve their stated objectives.

The Road to Obesity

To understand our present situation, let's step back and look at how we got to this point of a national health crisis. We believe that one major influence on our current predicament is the change that has occurred in our lifestyles. Think back fifty years ago—families generally consisted of two parents, and subsisted on one income. Family meals were prepared at home and enjoyed around the dinner

table. The newspaper was a major avenue for the flow of information, and businesses competed with the guy across the street.

Fast forward to the present—the composition of the family unit has changed, as well as the economic conditions in which it operates. Today, meals of convenience are the norm, and businesses conduct operations on a global scale. Information is transmitted as quickly as ideas are developed. The widespread use of cell phones, text messaging, and the internet have compounded the amount of information available to an individual at any given point in time. Consequently, the modern consumer expects instant satisfaction and greatly values added services and conveniences. Not surprisingly, the food industry has shifted toward providing indulgent, value-added food products that are highly convenient (see Capps and Park, 2003, for further discussion of food marketing channels). When you put this together with the facts that U.S. consumers generally have less discretionary time, more discretionary income, and lead sedentary lifestyles, you get a recipe for obesity.

In a continual effort to provide consumers with products they want, food marketers are watching these trends closely. Some recent new product trends emphasize the use of wholegrain ingredients, while others offer portion control like Nabisco's "100 Calorie Packs." Even so, marketers continue to struggle to increase per capita consumption of fresh fruit and vegetables, despite continued reports on the associated health benefits (Wang & McKay, 2006). However, the public outcry over the poor state of school food-service offerings may signal an opportunity for increased sales of fresh fruit and vegetables. In support of this, the government offers programs intended to improve the dietary intakes of school children while simultaneously supporting agricultural producers.

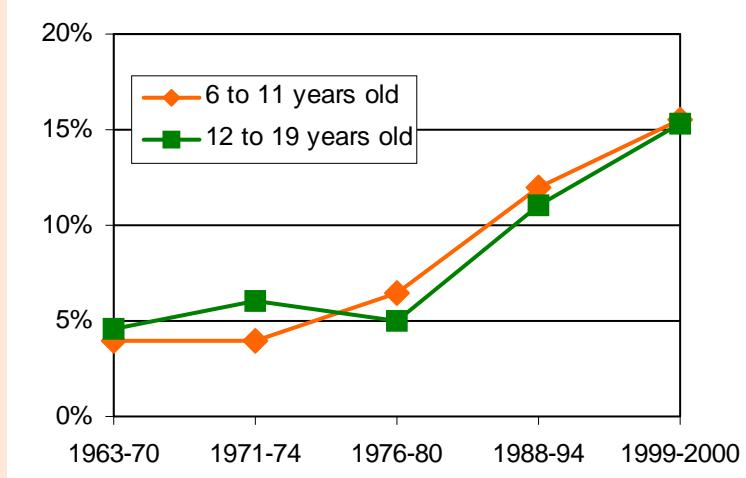


Figure 1. Percent of obese children.

Source: National Center for Health Statistics.

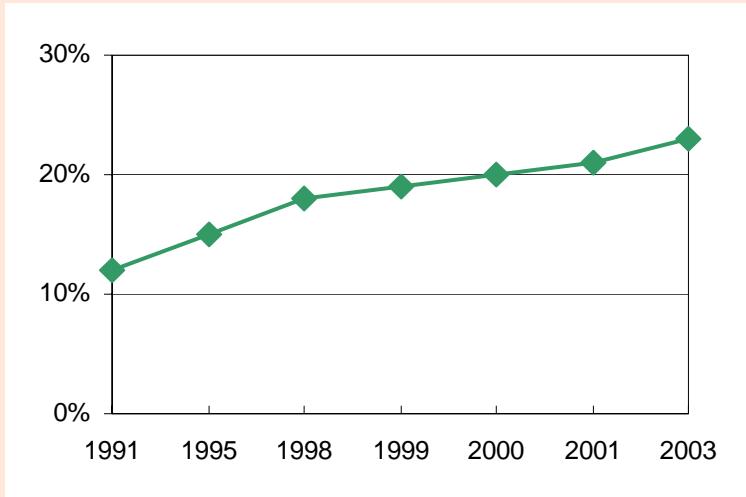


Figure 1. Percent of obese adults.

Source: National Center for Health Statistics.

Back to School

Most (if not all) school districts have a foodservice director that is in charge of purchasing food for the students within the district. Although their primary concern is providing lunch, many schools also offer breakfast and snacks. The foodservice director will combine funds available from state and local government as well as federal programs. In general, he/she can purchase products from

whatever source he/she chooses; however, participation in certain government programs requires purchasing specific products through specific sources of distribution.

A variety of programs are available to help foodservice directors procure food for their schools. Such programs include the National School Lunch Program (NSLP) and the National School Breakfast Program (NSBP) among others. The NSLP and NSBP differ from some

food aid programs in that they are available, at a slightly higher cost, to children who may not qualify for poverty-based assistance. The spending of these program funds are typically administered by a state department of agriculture.

The National School Lunch Program (NSLP) is the major government program that foodservice directors use to purchase their lunch foods. The NSLP provides nutritionally balanced low-cost, or sometimes free lunches to millions of children each school day. Since the inception in 1946, daily student participation in NSLP has grown from 7.1 million to 29.6 million in 2005, with approximately 100,000 schools participating. With regards to the NSBP, daily participation has grown from 1.8 million children in 1975 to 9.3 million children in 2005, with approximately 83,000 schools participating. Based on the large number of students using the NSLP and NSBP daily, their influence on nutrition, both in consumption and in establishing life-long behaviors, could be considerable.

There are also other programs that exist to encourage the consumption of specific food products in school programs. The Fresh Fruit and Vegetable Program, instituted by the USDA, reimburses schools for their purchases of fresh fruit and vegetables outside of those purchased as part of the NSLP. Initially available in 8 states, the program has been expanded, but funds are limited. For example, in Texas this program was made available to only 24 of the 7,203 schools that are eligible to participate in NSLP.

The methods school districts use to implement these programs go beyond putting nutritional foods on the menu. Some schools make these products available on demand,

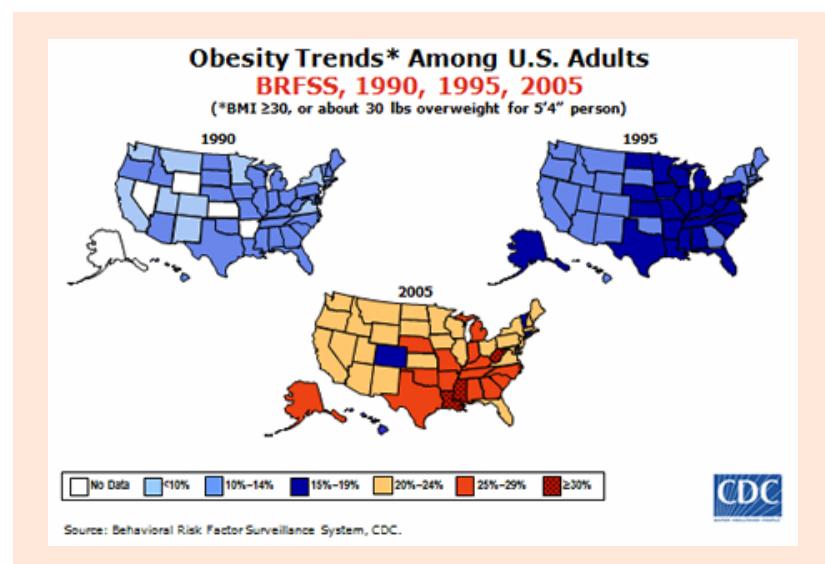
throughout the day. Finally, many states have initiated Farm-to-School programs in conjunction with federal programs. These programs help to keep federal funds within the state economy by allowing schools to buy produce from local growers at subsidized prices, sometimes only paying the cost of delivery (TDA, 2006).

Program Effectiveness

As part of the Centers for Disease Control and Prevention, the National Center for Health Statistics collects data through various methods in an effort to document the health status of the U.S. population. The information they gather is also an important part of research efforts to evaluate health policies and programs. However, quality of health is a complex issue. It can be measured in many different ways and is impacted by many different factors. For that reason, there is an abundance of research examining the effectiveness of these programs to provide only selected groups of nutrients at any one time.

Currently, we are examining data from the National Health and Nutrition Examination Survey (NHANES) to see if the NSLP and NSBP actually improve the consumption of fresh fruit and vegetables among school age children. Since obesity is rising and a large number of students eat at least one meal (lunch) and perhaps two meals (lunch and breakfast) at school each day, measuring the effectiveness of the NSLP and NSBP is extremely important in order to determine if the current guidelines are having an effect on healthy eating habits, particularly related to the consumption of fruits and vegetables.

Some preliminary results suggest that student participation in only the



NSLP has a positive impact on fresh fruit and vegetable consumption. However, student participation in the NSBP has a negative impact on fresh fruit and vegetable consumption (Campbell et al., 2007). Reasons behind these results are being investigated, but we need to remember that these results are influenced by the choices available to the students. For example, in the course of our research we were able to interview many different school foodservice directors. On one occasion, we ran across a reference to what foodservice personnel called "Hot Cheetos and cheese" that was sold to the students a la carte. The product involved taking a single serving bag of Flamin' Hot Cheetos (a popular brand of spicy extruded corn snack from Frito-Lay), pouring a scoop of melted nacho cheese over the contents, and putting a fork in it. This cheesy treat was a favorite among the students and provided the school district with sizeable revenue.

Although the product was admittedly unhealthy, the income that it generated gave the school district greater freedom and flexibility in operations. Any profit from the sale of a la carte items of this nature goes back to the district office, in essence

increasing its budget. The rare opportunity of an actual profit center in a school foodservice program is a temptation that can completely undermine nutritional objectives. This illustrates how the factors surrounding the implementation of foodservice programs can confound the ability of national programs to achieve their stated goals. Further, we found it interesting that smaller nearby school districts also admitted to selling the Flamin' Hot Cheetos and cheese mixture, but stopped that practice due to nutritional concerns. From discussions with these foodservice directors, it was evident that their action might be due in part to a greater sense of accountability to parents and increased parental involvement with school administrators.

As a final note, researchers need to be aware of the Cheetos effect. Seemingly, conflicting results surrounding federal program initiatives may not be entirely due to the program, but also due to the conditions of its implementation. National surveys sometimes have difficulty in accounting for quality differences among the experiences of their respondents. In order to be more effective, policy makers and food

marketers alike must be aware of the behavior of channel intermediaries like school administrators, in addition to the constraints they face.

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