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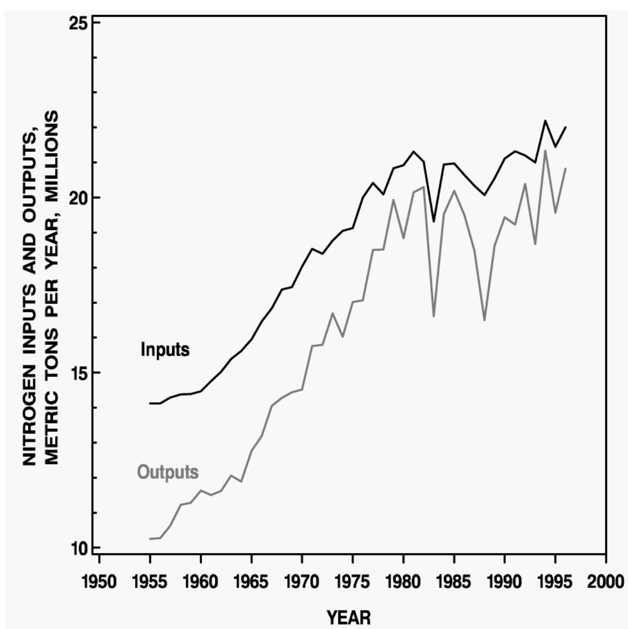
NOAA COASTAL OCEAN PROGRAM
Decision Analysis Series No. 20

Evaluation of the Economic Costs and Benefits of Methods for Reducing Nutrient Loads to the Gulf of Mexico

Topic 6 Report for the Integrated Assessment on Hypoxia in the Gulf of Mexico

Otto C. Doering, Francisco Diaz-Hermelo, Crystal Howard, Ralph Heimlich, Fred Hitzhusen, Richard Kazmierczak, John Lee, Larry Libby, Walter Milon, Tony Prato, and Marc Ribaudo

May 1999



U.S. DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
National Ocean Service
Coastal Ocean Program

GULF OF MEXICO HYPOXIA ASSESSMENT

This report is the sixth in a series of six reports developed as the scientific basis for an integrated assessment of the causes and consequences of hypoxia in the Gulf of Mexico, as requested by the White House Office of Science and Technology Policy and as required by Section 604a of P.L. 105-383. For more information on the assessment and the assessment process, please contact the National Centers for Coastal Ocean Science at (301) 713-3060.

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Cover image: Annual nitrogen inputs and outputs for the Mississippi–Atchafalaya River Basin from all major sources during 1951–96.

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May 1999

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Acknowledgments

We would like to thank Pat Dudley at Purdue University for her patience in receiving electronic bits and pieces from around the country and assembling them into this manuscript and doing this at all hours if necessary. The leaders of the other teams deserve recognition for their cooperative spirit in this venture. This Topic 6 report is the caboose of this train, and the cheerful assistance from the team leaders like Don Goolsby, Bill Mitsch, Andy Solow, and the others was absolutely essential in our ability to understand the problem, the science, and the trade-offs to get our job done. Our team members and additional contributing authors also maintained a sense of humor throughout, which allowed us to get this report completed in the 30-day time frame after the last scientific report was completed. Special credit and thanks go to the back room group at the Economic Research Service, individuals like Mark Peters and Roger Claassen who were always willing to pitch in and run the analytical models “just one more time.” They also made their models transparent and understandable to the rest of us, so we could judge how much confidence to place in the different results. We all borrowed heavily in terms of support from our home institutions as well, because we and they believed this was an effort to provide information for the public to make informed decisions and had to be done. Finally, the group at the National Oceanic and Atmospheric Administration, particularly Don Scavia and Mike Dowgiallo, created and defended our environment to do the highest-quality independent work possible. For this all the teams are grateful.

Otto Doering, for the Topic 6 Team
Purdue University

Foreword

Nutrient overenrichment from anthropogenic sources is one of the major stresses on coastal ecosystems. Generally, excess nutrients increase algal production and the availability of organic carbon within an ecosystem—a process known as eutrophication. Scientific investigations in the northern Gulf of Mexico have documented a large area of the Louisiana continental shelf with seasonally depleted oxygen levels (< 2 mg/l). Most aquatic species cannot survive at such low oxygen levels. The oxygen depletion, referred to as hypoxia, forms in the middle of the most important commercial and recreational fisheries in the conterminous United States and could threaten the economy of this region of the Gulf.

As part of a process of considering options for responding to hypoxia, the U.S. Environmental Protection Agency (EPA) formed the Mississippi River/Gulf of Mexico Watershed Nutrient Task Force during the fall of 1997, and asked the White House Office of Science and Technology Policy to conduct a scientific assessment of the causes and consequences of Gulf hypoxia through its Committee on Environment and Natural Resources (CENR). A Hypoxia Working Group was assembled from federal agency representatives, and the group developed a plan to conduct the scientific assessment.

The National Oceanic and Atmospheric Administration (NOAA) has led the CENR assessment, although oversight is spread among several federal agencies. The objectives are to provide scientific information that can be used to evaluate management strategies, and to identify gaps in our understanding of this complex problem. While the assessment focuses on hypoxia in the Gulf of Mexico, it also addresses the effects of changes in nutrient concentrations and loads and nutrient ratios on water quality conditions within the Mississippi–Atchafalaya River system.

As a foundation for the assessment, six interrelated reports were developed by six teams with experts from within and outside of government. Each of the reports underwent extensive peer review by independent experts. To facilitate this comprehensive review, an editorial board was selected based on nominations from the task force and other organizations. Board members were Dr. Donald Boesch, University of Maryland; Dr. Jerry Hatfield, U.S. Department of Agriculture; Dr. George Hallberg, Cadmus Group; Dr. Fred Bryan, Louisiana State University; Dr. Sandra Batie, Michigan State University; and Dr. Rodney Foil, Mississippi State University. The six reports are entitled:

> ***Topic 1: Characterization of Hypoxia.*** Describes the seasonal, interannual, and long-term variations of hypoxia in the northern Gulf of Mexico and its relationship to nutrient loadings. *Lead: Nancy N. Rabalais, Louisiana Universities Marine Consortium.*

Topic 2: Ecological and Economic Consequences of Hypoxia. Evaluates the ecological and economic consequences of nutrient loading, including impacts on the regional economy. *Co-leads: Robert J. Diaz, Virginia Institute of Marine Science, and Andrew Solow, Woods Hole Oceanographic Institution, Center for Marine Policy.*

Topic 3: Flux and Sources of Nutrients in the Mississippi–Atchafalaya River Basin. Identifies the sources of nutrients within the Mississippi–Atchafalaya system and Gulf of Mexico. *Lead: Donald A. Goolsby, U.S. Geological Survey.*

Topic 4: Effects of Reducing Nutrient Loads to Surface Waters Within the Mississippi River Basin and Gulf of Mexico. Estimates the effects of nutrient-source reductions on water quality. *Co-leads: Patrick L. Brezonik, University of Minnesota, and Victor J. Bierman, Jr., Limno-Tech, Inc.*

Topic 5: Reducing Nutrient Loads, Especially Nitrate–Nitrogen, to Surface Water, Ground Water, and the Gulf of Mexico. Identifies and evaluates methods for reducing nutrient loads. *Lead: William J. Mitsch, Ohio State University.*

Topic 6: Evaluation of the Economic Costs and Benefits of Methods for Reducing Nutrient Loads to the Gulf of Mexico. Evaluates the social and economic costs and benefits of the methods identified in Topic 5 for reducing nutrient loads. *Lead: Otto C. Doering, Purdue University.*

These six individual reports provide a foundation for the final integrated assessment, which the task force will use to evaluate alternative solutions and management strategies called for in Public Law 105-383.

As a contribution to the Decision Analysis Series, this report provides a critical synthesis of the best available scientific information regarding the ecological and economic consequences of hypoxia in the Gulf of Mexico. As with all of its products, the Coastal Ocean Program is very interested in ascertaining the utility of the Decision Analysis Series, particularly with regard to its application to the management decision process. Therefore, we encourage you to write, fax, call, or e-mail us with your comments. Our address and telephone and fax numbers are on the inside front cover of this report.

David Johnson, Director
Coastal Ocean Program

Donald Scavia, Chief Scientist
National Ocean Service

Executive Summary

In this report we analyze the Topic 5 report's recommendations for reducing nitrogen losses to the Gulf of Mexico (Mitsch et al. 1999). We indicate the relative costs and cost-effectiveness of different control measures, and potential benefits within the Mississippi River Basin. For major nonpoint sources, such as agriculture, we examine both national and basin costs and benefits.

Based on the Topic 2 economic analysis (Diaz and Solow 1999), the direct measurable dollar benefits to Gulf fisheries of reducing nitrogen loads from the Mississippi River Basin are very limited at best. Although restoring the ecological communities in the Gulf may be significant over the long term, we do not currently have information available to estimate the benefits of such measures to restore the Gulf's long-term health. For these reasons, we assume that measures to reduce nitrogen losses to the Gulf will ultimately prove beneficial, and we concentrate on analyzing the cost-effectiveness of alternative reduction strategies. We recognize that important public decisions are seldom made on the basis of strict benefit–cost analysis, especially when complete benefits cannot be estimated. We look at different approaches and different levels of these approaches to identify those that are cost-effective and those that have limited undesirable secondary effects, such as reduced exports, which may result in lost market share.

We concentrate on the measures highlighted in the Topic 5 report, and also are guided by the source identification information in the Topic 3 report (Goolsby et al. 1999). Nonpoint sources that are responsible for the bulk of the nitrogen receive most of our attention. We consider restrictions on nitrogen fertilizer levels, and restoration of wetlands and riparian buffers for denitrification. We also examine giving more emphasis to nitrogen control in regions contributing a greater share of the nitrogen load.

Although we are limited by existing data and existing analytical capacity, within these constraints we provide information for making policy judgments by setting bounds and parameters for different approaches to nitrogen reduction. Topic 5 was primarily concerned about producers' ability to achieve nitrogen reductions using feasible production practices. Our analysis accounts for economic impacts on the producers and keeps changes in acreage and exports within historic bounds of recent past adjustments—something of concern to many in the agricultural sector.

Fertilizer restrictions are a more cost-effective means of reducing nitrogen losses than strategies based only on wetland restoration or buffers. They are more cost-effective than a fertilizer tax, because of the tax's impacts on producer net returns. Wetland-based strategies are more expensive than fertilizer-reduction strategies to achieve the same goal of reducing nitrogen loss. Land-retirement costs and wetland-restoration costs outweigh the higher environmental benefits generated by wetlands. Based on uniform assumptions about denitrification efficiency, focusing on restoring wetlands proportional to nitrogen losses is less cost-effective than enrolling wetlands at lowest cost. Vegetative buffers are least cost-effective, due to low nitrogen filtering relative to wetlands, lower wildlife-associated benefits, and high land-retirement costs.

A 5-million-acre wetland restoration combined with a 20% reduction in fertilizer is the most cost-effective, practicable strategy we examined for meeting a 20% nitrogen loss-reduction goal. This strategy reduces nitrogen loss by about 20% with few, if any, secondary effects that are beyond our historical experience of sectoral adjustment in agriculture. Reducing fertilizer by 45% meets the goal for a slightly higher cost. A policy that includes wetlands has additional advantages because it meets other policy objectives and generates wildlife and recreation benefits.

For the agricultural sector, cost savings from reduced fertilizer nutrient inputs are modest in most cases. However, commodity prices and aggregate producer net returns rise at increasing levels of nitrogen-loss reductions. This is not a result of lower nitrogen fertilizer costs; instead, it derives from reduced production resulting from reduced fertilizer inputs. These begin to be significant when nitrogen-loss restrictions reach 30% and higher. Aggregate returns to U.S. agriculture increase, but costs are imposed on some who are constrained to abandon profitable production in order to meet nitrogen-loss goals. Severe restrictions on nitrogen loss from agriculture mean that production ceases on acres in the Mississippi River Basin that are especially vulnerable to nitrogen loss. The restrictions also cause shifts to cropping systems that lose less nitrogen. Production of crops with high nitrogen losses is also increasingly shifted out of the basin. Some producers suffer these losses, while those remaining in production with cropping systems that provide relatively high value reap benefits from increased commodity prices as the supply is reduced due to nitrogen restrictions.

We find only modest aggregate impacts on the sector for up to a 20% nitrogen-loss reduction (comparable to the 15–20% reduction in nitrogen losses from agriculture deemed feasible and recommended by the Topic 5 team). We find that restoring 5 million acres of former wetlands also has minimal impact on agricultural production and related factors. At the 10-million-acre level, noticeable price, land-use, and other impacts occur.

Livestock producers bear more costly feed grain input costs as prices increase under nitrogen-loss restrictions. Consumers of basic commodities, and the finished food and fiber products derived from them, suffer some loss from price increases caused by production changes and acreage restored to wetlands. There is also a potential cost from decreased agricultural export volumes that depends upon the level of nitrogen restriction (although the *value* of exports increases because of price increases). Export reductions become more important and begin to break out of historical bounds when nitrogen loss is restricted to 30% or more. The primary concern of the agribusiness industry is loss of sales in an expanding free-market environment where market share is voluntarily constrained to meet environmental objectives. Also, reduced acreage in production and reduced output can have negative impacts on input and shipping sectors.

As nitrogen use is restricted inside the basin, increased nitrogen, phosphorus, and sediment loss in agricultural production occurs outside. Reducing nitrogen losses to the Gulf is likely to impose additional pollution costs on the rest of the nation as an indirect impact. Price increases due to reduced production within the basin will intensify crop production elsewhere. The extent of derived environmental impacts is estimated to be up to the 20% nitrogen loss-reduction level.

Finally, institutional factors are important in any broad-based effort to reduce nitrogen loss in the Mississippi River Basin. For any program, administration, monitoring, verification, and enforcement costs and capabilities must play an important part in the final choice of strategy or action. These costs become even more critical in a region such as the Mississippi River Basin, which includes many independent political jurisdictions. Policies need to be coordinated across political boundaries, and the costs of coordination increase if multiple strategies are employed.

CHAPTER 1

Introduction

1.1 SCOPE OF THIS REPORT

The objectives assigned to the Topic 6 team were to:

> "Evaluate the social and economic costs and benefits of the methods identified in Topic 5 for reducing nutrient loads. This analysis will include an assessment of various incentive programs and will include any anticipated fiscal benefits generated for those attempting to reduce [nitrogen] sources."

We compiled the information available in the literature and analyzed it where analytical tools and data already existed. In carrying out our task, we have:

- Analyzed the Team 5 recommendations as far as possible.
- Used the Topic 2 analysis of the costs of hypoxia to the Gulf to represent the value of benefits that may be ascribed to the Gulf from reducing nitrogen flows from the Mississippi Basin to the Gulf.
- Identified the relative costs of reducing nitrogen flows, to indicate the cost-effectiveness of various measures.
- Indicated, to the best of our ability, estimates of consumer and producer surplus, tax burdens, and incidence of costs to various groups.

Although we lacked an adequate foundation of existing work to estimate social costs, we have kept such costs in mind and have sometimes flagged them in analyzing alternative ways of reducing nutrients that would create relatively greater or lesser social costs.

For this report we have not recommended or analyzed specific policies or policy alternatives. These will be considered in the Integrated Assessment that will draw together the work of all of the six reports. Whenever possible, we have analyzed different actions that may be taken to reduce nitrogen flows and have presented ranges of possible actions to allow the Integrated Assessment team to judge the efficacy, secondary impacts, and cost-effectiveness of a particular action applied at different levels.

1.2 REPORT OUTLINE

Chapter 2 refers to the work of the Topic 3 team on the sources of nitrogen and their magnitudes, proportional contributions, and characteristics. It reviews some of the important guidance developed by the Topic 5 team on setting priorities and concentrating on specific sectors and approaches. It also presents some important aspects of existing policy-setting strategies, along with background information on existing work presented for important focus points in the Topic 3 and Topic 5 reports, including nitrogen loss from agriculture and economic studies on wetland use in nitrogen control.

Chapter 3 provides a background for the assumptions and criteria that are common to economic analysis. It explains how the characteristics of nonpoint-source pollution are particularly important to analyzing potential mitigation policies because most of the nitrogen reaching the Gulf comes from nonpoint sources. This chapter also discusses the economic considerations important in selecting policy instruments, examines some of our criteria and assumptions, and briefly looks at benefit–cost analysis and an explanation of our criteria and assumptions.

Chapter 4 analyzes the costs to the agricultural sector and the rest of society of both using alternative strategies in the Mississippi Basin to reduce nitrogen loss from agriculture and using different amounts of wetlands to control nitrogen loss. Different ranges of control are simulated for the two aggregative analyses to assess indirect and well as direct costs and impacts. The chapter also includes a discussion of point-source reductions.

Chapter 5 looks at the environmental benefits within the basin resulting from different actions taken to reduce nitrogen loss to the Gulf of Mexico. Chapter 6 assesses the results and identifies the strategies that achieve established program goals at least cost. Chapter 7 discusses institutional considerations relevant to a range of different policy options, and Chapter 8 summarizes this report’s important findings.

Appendices A and B provide information on and the results of an analysis of animal waste and atmospheric deposition. Appendix C describes the EPIC (Erosion Productivity Impact Calculator) model used within the U.S. Mathematical Programming system.

CHAPTER 2

Problem Setting and Methods

2.1 SOURCES OF NITROGEN

The nitrogen balance table developed in the Topic 3 report provided a critical input for the Topic 5 and Topic 6 analyses. The relative contributions of direct nitrogen inputs and recycled nitrogen inputs are presented in Table 2.1. To the extent possible we have been guided by the balance sheet for our emphasis on various sources.

TABLE 2.1. Annual nitrogen inputs.

Sources of Nitrogen	Percent Input
Nonpoint Sources	
New Inputs	
Fertilizer	30.0
Legumes and pasture/hay	19.0
Atmospheric deposition	5.5
Recycled Inputs	
Potentially mineralizable soil nitrogen	29.5
Manure	12.0
Wet ammonia deposition	2.4
Point Sources	
Municipal	1.0
Industrial	0.3
Total	@100.0

2.1.1 Guidance from Topic 5

Most of our effort has been on nonpoint sources, reflecting the recommendations from the Topic 5 report on methods and impacts of reducing nonpoint sources. This includes nitrogen losses from agricultural fertilizer, legumes and pasture, manure, and potential mineralization of soil nitrogen. Together these account for 90.5% of the total.

We have not been able to account for manure as being explicitly separated out from other nonpoint nitrogen sources. Manure is also a difficult balance sheet variable because corn produced in one place with fertilizer inputs may be fed through an animal in the same place, and the manure nitrogen thus may be subject to double counting. However, we have added a more extensive discussion of manure as a source because of the current concerns about manure as a pollutant and because we believe that changes in the structure of the animal industry can lead to a greater proportion of the manure in the basin ending up in the Gulf (see Appendix A).

We have presented some of the information available on atmospheric deposition of nitrogen. This comes primarily from Chesapeake Bay, but provides some estimates of costs of reducing atmospheric nitrogen. We have also provided some analysis on municipal point-source nitrogen through an example of using tradable permits to reduce the cost of more stringent point-source requirements (see Appendix B).

The Topic 5 report highlighted four approaches to reducing nitrogen loadings:

- Reducing nitrogen use by and nitrogen loss from agriculture.
- Intercepting laterally moving water through riparian buffers, controlled drainage, and wetlands, particularly targeting areas with high concentrations of nitrates.
- Installing tertiary treatment systems for point sources.
- Providing a system of river-diversion backwaters in the Mississippi Delta and Upper Mississippi.

To a great extent our analysis of reducing agricultural nitrogen losses parallels the Topic 5 recommendations in adjusting fertilizer levels, changing practices and cropping systems. We did not analyze expanding the distance between tile lines. We also approached the animal manure source differently, looking at the major component that is spread on the land.

We have analyzed the restoration of wetlands and riparian buffers in the amounts discussed in the Topic 5 report. We have not analyzed controlled drainage, which is much more limited in application. We also did an analysis of concentrating the wetlands and buffers in regions of high nitrogen concentrations.

Our approach to tertiary treatment of point sources has been to look at the extent to which direct treatment costs could be mitigated by trading with less costly nonpoint-source control—in this case agriculture. The suggestion to use wetlands for treatment can be considered on the basis of the acres required and the costs and benefits of wetlands from our specific wetlands analysis.

Both the river diversion at the Delta and flood diversion in the Upper Mississippi relate again to the creation of additional areas of wetlands plus additional engineering works. These were not analyzed as a specific case, but would be based on the analysis of the benefits and costs of wetland restoration.

2.1.2 Geography of Nitrogen Sources

The Topic 3 report maps those watersheds contributing the highest concentrations of nitrogen. On an engineering basis and on the basis of cost per unit of nitrogen reduced, one normally thinks in terms of first reducing the pollutant from the most concentrated sources. However, we found this approach was not as clear-cut for nonpoint sources, given the limited information we had. From the Topic 3 maps, the upper Corn Belt is an area of high concentration. From our analysis of reducing the net loss of nitrogen from agriculture across the basin, one can see the extent to which this area makes more adjustments—for example, in fertilizer use and cropping shifts—to achieve a given reduction in nitrogen loss with the least financial loss to farmers. In creating wetlands to reduce nitrogen loss, we did examine concentrating new wetlands geographically on the “hot spots” of nitrogen loss; however, we found no clear advantage in doing so. The geographical limitation to a hot-spot watershed or region tended to greatly increase land acquisition cost, given the limited supply of wetland sites.

2.2 POLICY SETTING

2.2.1 Water Quality Laws

2.2.1.1 THE CLEAN WATER ACT

In fall of 1997, Vice President Gore directed the U.S. Environmental Protection Agency (EPA) and the U.S. Department of Agriculture (USDA) to collaborate in preparing a Clean Water Action Plan to implement the Clean Water Act of 1972, as amended. The general goal of that parent legislation is to achieve “fishable and swimmable waters” for all Americans. Twenty-six years later, much remains to be done. Approxi-

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ERRATUM SHEET

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May 1999

The back of this page provides revised text for the first paragraph of Section 2.2.1.1 The Clean Water Act

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In fall of 1997, Vice President Gore directed the U.S. Environmental Protection Agency (EPA) and the U.S. Department of Agriculture (USDA) to collaborate in preparing a Clean Water Action Plan to implement the Clean Water Act of 1972, as amended. The general goal of that parent legislation is to achieve "fishable and swimmable waters" for all Americans. Twenty-six years later, much remains to be done. Approximately 40%, or 18,000, of those water bodies tested are still out of compliance with that goal. A 1994 report to Congress (USEPA, 1994a) indicated that 23% of river impairments, 43% of lake impairments, and 47% of estuarine impairments were caused by nutrient enrichment. Two years later, the 1996 *National Water Quality Inventory* (USEPA 1998a) reported even higher levels of nutrient impairment: 40% of impaired rivers, 51% of impaired lakes, and 57% of impaired estuaries. Agriculture was identified as the most widespread source of pollutants, followed by municipal sewer systems and urban storm-water runoff. While point sources have been largely controlled, nonpoint pollution from agricultural, suburban, and urban sources remains the most challenging national water quality problem.

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The Clean Water Action Plan, prepared jointly by USDA and EPA and released by President Clinton in February 1998, calls for extensive collaboration within the states to deal with the nonpoint problems. There is to be a substantial increase in technical and financial support for state and local efforts. States are required to implement nondegradation policies, and EPA will work with the National Oceanic and Atmospheric Administration (NOAA) in coastal states to develop additional enforcement authority in case the voluntary/incentive-based approach to nonpoint abatement is not effective.

The Plan also calls for improved standards and criteria for defining water quality problems and gauging progress. An initial cut at those standards lists seven priorities: strengthening ambient water quality criteria, developing nutrient standards, developing specific standards for microbial pathogens, completing biocriteria for aquatic life, improving methods for measuring and achieving total maximum daily loads (TMDLs), considering possible criteria for sediment and flow characteristics, and finding ways to implement these standards and criteria throughout the U.S. (USEPA 1998c). EPA is making a strong effort to collaborate with state and local agencies and to involve water quality stakeholders in this entire process. The standards and their implementation must acknowledge differences among states and regions of the country. National “guidance documents” are being prepared to identify techniques for measuring the trophic state of water bodies and establishing appropriate nutrient criteria for improvement. These are to be available by 2000, with the expectation that states will have their own criteria in place by 2003. EPA will assist in the state process by sharing with all the states the information it receives from individual states.

Another important part of the Clean Water Action Plan establishes goals for reducing pollution from animal feeding operations. A draft "Unified National Strategy for Animal Feeding Operations" was released by EPA Administrator Browner and USDA Secretary Glickman on September 16, 1998. These animal operations create nutrient problems for 35,000 miles of nearly 700,000 miles of river surveyed, including segments that feed the Mississippi and eventually the Gulf of Mexico. The strategy emphasizes voluntary action by livestock producers to develop comprehensive nutrient management plans by 2008. Units larger than 1,000 animals and those discharging directly into water bodies will be required to develop such plans as part of EPA's current permitting process. EPA will also be reviewing national environmental guidelines for all animal operations. (See Appendix A for a further discussion of the animal waste issue.)

2.2.2 Conservation Policy Setting

Several important federal laws establish the context for further actions to reduce nutrient pollution of the Gulf of Mexico. Farmers make production and marketing decisions in response to incentives established in markets that are defined by various rules for participation, including those contained in federal law. Changes in farmer behavior in the interest of further improving downstream water quality may require adjustments in those market rules. Policies and programs affecting water quality have emerged at all levels of government, but of most importance here are federal laws that transcend local and state boundaries. Particularly important are the federal conservation programs that are usually included in the various farms bills. The Conservation Reserve Program and the Environmental Quality Incentive Program, described below, are the largest of these.

In the future, states are likely to take on an increasingly important role in protecting water quality. Initially, this will be in the form of regulation and special restrictions, like Iowa's and North Carolina's regulations relating to livestock operations animal waste. Many states are unlikely to be willing to spend the dollars targeted toward conservation and water quality that the federal government does. However, it appears likely that states will adopt a regulatory approach toward specific statewide or regional problems that will be more constraining than the blanket federal regulation.

2.2.3 Agricultural Policy Setting

2.2.3.1 FARM AND FOOD POLICY

The "greening" of U.S. food policy really began with the Food Security Act of 1985 and has continued through the current Federal Agricultural Improvement and Reform Act of 1996 (FAIR). Most likely, any future farm and food legislation will also acknowledge the relationships between food production and natural resource quality.

Title XII of the 1985 Farm Law (P.L. 99-198) introduced Sodbuster, Swampbuster, and conservation compliance provisions to establish a firm policy link between the price- and income- support aspects of food policy and protecting the quantity and quality of natural resources. Farmers could retain eligibility for income supports only by protecting natural resources. The success of the environmental incentives depended very much on the availability and attractiveness of the income supports to eligible farmers (Reichelderfer 1990b). The Conservation Reserve Program (CRP) was introduced to permit government to lease the most erosive lands for 10 years to protect them against further damage or to prevent them from contributing to water quality problems downstream. Primary emphasis in 1985 was on-farm soil productivity, rather than off-farm damages.

The Food, Agriculture, Conservation and Trade Act of 1990 (P.L. 101-624) continued the green initiatives of the 1985 law, by adding the watershed-based Water Quality Incentive Program and expanding the CRP to focus more on off-farm water quality than on soil erosion. There has been much concern about the expiration of the temporary CRP contracts and the potential consequences (Ervin et al. 1991). All of these programs rely on market incentives to encourage a pattern of farmer decisions that will have attractive social consequences. The cross-compliance provisions, however, added a mandatory aspect by requiring farmers to recognize that if they are to enjoy the benefits of income protection or risk reduction by government programs, they must consider the impacts of their production decisions on other natural resource users.

The 1996 Farm Law (P.L. 104-127) further strengthened the environmental thrust of the conservation reserve with the Environmental Conservation Acreage Reserve Program (ECARP), and added an important incentive program to encourage farmers to reduce runoff that causes water quality problems (Ogg and Kuch 1997). Deliberate language to “reconcile productivity and profitability with protection and enhancement of the environment” clearly establishes the intent of this new era in farm and food policy. The law continues conservation compliance but grants farmers additional flexibility in establishing compliance.

While ECARP enables some farmers to terminate CRP contracts early, those provisions do not apply to lands that have an erodibility index greater than 15 and that include filter strips, grass waterways, or riparian areas. Farmers may also sell long-term or permanent easements on wetlands and undertake wetland restoration with cost-share assistance through USDA (Osborn 1996). While the link between wetland easements and eutrophication problems in the Gulf of Mexico may be indirect, wetlands perform critical environmental services in conjunction with farm operations that have long-term significance.

Two programs that provide incentives for restoring wetlands and riparian buffers to intercept nutrients before they reach streams are the Wetland Reserve Program (WRP) and USDA’s Conservation Buffer Initiative. WRP, first authorized in the 1990 farm bill and continued in the 1996 FAIR Act, has authority to enroll 975,000 acres of cropland that was formerly wetland in long-term or permanent easements and to share the cost of restoring wetlands (USDA 1997). Along with the Emergency Wetlands Reserve Program (EWRP), more than 700,000 acres of restored wetlands have been enrolled to date, with a large proportion in the Mississippi Delta and Corn Belt regions. The Conservation Buffer Initiative builds on efforts in several programs, primarily the continuous sign-up provisions of the Conservation Reserve Program and the Conservation Reserve Enhancement Program (CREP) (USDA/NRCS 1997). Producers willing to restore riparian buffers to permanent grass or trees can bypass competitive bidding in regular sign-up periods. Annual rental costs and a share of the restoration costs are paid. The CREP uses the continuous sign-up in conjunction with additional state program incentives to encourage buffer restoration. More than 700,000 acres of continuous sign-up practices have been enrolled to date, and USDA has approved CREPs in seven states, with nearly a dozen more in the application process.

The Environmental Quality Incentive Program (EQIP) of the 1996 Act is directly aimed at inducing farmers to do a better job of reducing nutrient runoff than they would otherwise. It is meant to push them beyond what may be rational for their business and not simply pay for what they are inclined to do anyway (Libby 1998). For reasons more political than scientific, half of the EQIP dollars are to be targeted on livestock operations smaller than 1,000 animal units.

While the program language speaks of “maximizing environmental benefits per dollar spent” at the national level, allocation priorities are set by states, with national efficiency defined as the sum of state priorities as further influenced by the 50% mandate for livestock. Farmers bid for EQIP dollars by indicating their 5- to 10-year conservation plans, which include changes to cropping systems, manure, and nutrient management. Conservation tillage options are seen as particularly effective in reducing nitrate and phosphate runoff (Fawcett 1995). Total incentive and cost-share payments for conservation and resource protection may not exceed $10,000 per farm per year, or $50,000 for the full contract. Total authorization is $200 million per year, most of which is redirected from the Commodity Credit Corporation (Osborn 1996). EQIP payments are included with other positive programs in the farm legislation as incentives for the farmer under the Conservation Compliance Program noted above.

Other provisions of the FAIR Act relevant to Gulf of Mexico water quality implications of agriculture include the Forestry Incentives Program, authority for flood plain easement purchase, the Wildlife Habitat Incentive Program, and the Farmland Protection Program, which helps state and local governments buy conservation easements on farmland.

2.3 REDUCING NITROGEN LOSS FROM FERTILIZER

Fertilizer is a major factor in the nonpoint nitrogen loss from agriculture. One reason for this is the extremely high value resulting from the use of nitrogen. For example, at county average levels of corn production, an extra pound or pound and a half of nitrogen will yield an extra bushel of corn if other nutrients and moisture are adequate. At $0.18 a pound, nitrogen can yield an extra bushel worth $2 or more.

There are two ways farmers react to this point strategically:

- First, extra nitrogen may be supplied as insurance against nitrogen loss that would cut production. Such loss could occur with heavy rainfall, saturated soil, and resulting denitrification. An extra 10–20 pounds of nitrogen costing a few dollars can result in an extra 10–15 bushels worth at least $20–$30. This is a very rational insurance approach to adding what may otherwise be excess nitrogen.
- Second, farmers may add extra nitrogen to take advantage of an especially good year when moisture and other nutrients are not limiting, temperature is just right, and extra nitrogen will give a boost to production. Again, an extra couple of dollars have the capacity to increase returns tenfold in an especially good year as well as in a bad year when nitrogen may be lost.

These trade-offs are illustrated by the case studies in section 2.3.1, which explore the economics of reducing nitrogen loss. Unfortunately, there are few case studies that examine these issues in a thorough, consistent way that can provide guidance across the entire Mississippi Basin. While these case studies provide some insight, a consistent modeling approach, such as that provided by the U.S. Mathematical Programming (USMP) modeling framework (which is supported by biophysical process modeling of nitrogen losses) is required to make credible estimates of both the economic and the physical impacts of alternative approaches to reducing nitrogen from agricultural production.

2.3.1 Reducing Nitrogen Loss from Agriculture

Qiu (1996) evaluated the economic and environmental impacts of alternative farming systems and trade-offs between watershed net returns and nonpoint-source pollutants in Missouri's Goodwater Creek watershed. Because any case study reflects the soil, climate, and physiographic characteristics of its location, it cannot be widely extrapolated to an area as diverse as the entire Mississippi River Basin. Goodwater Creek watershed is located in a claypan soil region that covers a swath from northeastern Oklahoma to southwestern Illinois, including portions of Missouri; therefore, the results for Goodwater Creek watershed cannot be generalized widely.

In Goodwater Creek, annual net return per hectare was directly related to fertilizer application rate when other factors were held constant. High fertilizer and pesticide applications distinguished the farming system with the highest returns over all sub-watersheds ($237.65/ha, or $96.17/ac) from the least profitable farming system ($49.13/ha or $19.88/ac) with the same tillage and rotations.

Concentration of nitrate–nitrogen in surface runoff was significantly affected by fertilizer application rate and crop rotation, and varied spatially in the watershed. Farming systems with higher nitrogen application rates generated higher losses, other factors held constant. Within the same fertilizer application category (high, medium, or low), losses were highest for farming systems with more row crops (corn and soybeans) in the rotation and lowest for rotations with more close-grown crops, such as wheat. Farming systems with high fertilizer application rates that had more row crops in the rotation generated the highest nitrogen losses. Average concentrations in runoff for these farming systems were 13.45 ppm without riparian buffers, and 3.82 ppm with riparian buffers, reducing surface losses by about 70%.

Qiu found significant trade-offs between profitability and water quality. Total watershed net return decreased as nitrogen losses were decreased. Without riparian buffers, total watershed net return decreased $26,483 per ppm for a 5% reduction in losses and $37,298 per ppm for a 50% reduction. Total net return decreased more in some sub-watersheds than in others as water quality improved. For certain sub-watersheds, there was no trade-off between total net return and water quality.

2.3.1.1 NITROGEN CREDITING AND TESTING

Economic theory tells us that in dealing with the problem of nonpoint-source pollution—particularly nitrogen (N) contamination of surface water—the most efficient strategy is to reduce N application rates in areas where they are excessive for crop needs. This is essentially a win-win situation because reducing N application rates both reduces fertilizer costs and increases profit margins, as well as decreases N contamination of surface water. This win-win situation can only occur if producers are operating on the flattened portion of the fertilizer response function. While farmers may be optimizing physical production rather than economic return, experience and good agronomic extension knowledge will have moved them to the left, back toward the portion of the yield-response function in which reducing N application rates implies a reduction in crop yields. Except within a relatively narrow range of reductions, decreased N application will reduce crop yields in this range and reduce incomes.

Most crop production today is based on general, soil- or region-based fertilizer recommendations developed by university agricultural extension personnel. Nutrient planning based on crediting all potential sources of nitrogen and testing soils, plants, and manures for nitrogen content can reduce nitrogen applications over typical practice. Nitrogen management can be improved by increasing the efficiency of nitrogen use, defined as the percent of N applied to the land that is used by plants (Mabler and Bailey 1994). Proper crop and N fertilization management can reduce nitrate loss to the environment and achieve optimum crop production (Keeney and Follett 1991). Recent evaluations of long-term corn experiments show that fertilizer N removal by corn grain rarely exceeds 40% of total available N, and is often much less at economically optimum corn yields (Blackmer 1986; Oberle and Keeney 1990). Depending on the initial level of fertilization, efficiency can be improved by increasing crop uptake of applied N, achieving the same or higher yield with reduced application of N, and reducing N losses by changing the timing and/or method of application (Bock and Hergert 1991).

The extent to which producers across the Mississippi River Basin are currently overfertilizing is unknown, but some case studies indicate that significant reductions can occur without reducing yields. Based on USDA Economic Research Service surveys of Nebraska farmers, Fuglie and Bosch found that nearly half of the surveyed farmers have used N fertilizer recommendations from a preplant N test and were achieving N fertilizer reductions of 18–33% with no loss in yield. Shortle et al. (1994) found that 36% of farmers used late spring soil tests and were able to reduce N fertilizer use by 40%. A study of USDA cost and returns data by Trachtenberg and Ogg found that N fertilizer savings of 24–32% could be obtained by crediting all sources of N available on the farm.

While some reduction in N fertilizer over typical application rates could be obtained using crediting and preplant soil testing, the costs of providing this information to producers and providing sufficient incentive to ensure adoption of these methods are not well known.

2.3.1.2 PRECISION NITROGEN APPLICATION

Applying nitrogen at rates that exceed crop uptake can increase nitrate–N concentrations in surface and ground water, contaminate drinking-water supplies, and degrade aquatic ecosystems. Nitrogen fertilizer is typically applied to a field at a uniform rate. Application rates needed to achieve economically optimum crop yields, however, can vary within fields due to spatial variability in soil moisture, soil N mineralization rates, and the efficiency with which crops use N. Uniform application of N may not achieve maximum net return when N is overapplied in some areas and underapplied in other areas of a field. Overapplication of N could degrade water quality, and underapplication could reduce crop yield and net return. Varying the N application within a field based on site-specific growing conditions can reduce over- or underapplication of fertilizers (Kitchen et al. 1992) and increase the efficient use of N (Fiez et al. 1994; Sawyer 1994).

Prato and Kang (1998) evaluated the economic and environmental impacts of variable and uniform N application for 35 sample fields in Goodwater Creek watershed. First, a soil type analysis evaluated the differences across the 10 soil types in the 35 sample fields. Second, a field-level analysis evaluated the differences across the 35 fields.

On the same soil type, variable application generally produced higher returns, but often also led to higher nitrogen and phosphorus losses in runoff. These differences were more pronounced for rotations of row crops (corn and soybeans), and less likely with rotations involving sorghum and wheat, where uniform application was more profitable. These results generally held when comparing across fields composed of different soil types, as well. Overall, variable-rate application increased both net return and nutrient losses to the environment for row-crop rotations, but produced mixed or negative results with close-grown crops.

The study results indicated that the profitability and water quality benefits of variable application are sensitive to the distribution of soil types in a field and with crop rotation. Despite the intuitive logic of matching nitrogen application to site-specific crop needs, variable application was not uniformly superior to uniform application in terms of increasing net return and improving water quality in Goodwater Creek watershed.

2.3.2 Restoring Wetlands to Reduce Nitrogen Loss

Case studies of the economics of wetland restoration have generally focused on estimating the costs, rather than the benefits, of restoration. They provide some guidance for modeling wetland restoration, but are not comprehensive enough to substitute for a systematic modeling approach.

Heimlich et al. (1989) and Carey et al. (1990) determined that the average easement and restoration costs for a least-cost wetland reserve from restoring hydric cropland ranged from $845 million for a 1-million-hectare (2.5-million-acre) reserve ($845/ha or $341.95/ac) to $2.4 billion for a 4-million-hectare (10-million-acre) reserve ($600/ha or $243/ac) in 1988 dollars. Minnesota, Iowa, and Missouri would have the highest wetland reserve acreage. Several studies estimated the present value of net returns from converting wetland to agricultural land, including land-clearing and -preparation costs. These values can be viewed as the opportunity cost (loss in net agricultural income) of restoring agricultural land to wetlands. Present values ranged from $376/ha ($152/ac) in the Mississippi Delta region (Kramer and Shabman 1986), to $1,573/ha ($637/ac) in North Carolina (Danielson et al. 1988; Danielson and Hamilton 1989), to $635/ha ($257/ac) in central Minnesota (Danielson and Leitch 1986).

The Des Plains River Wetland Demonstration Project in Wadsworth, Illinois, evaluated the economic efficiency and political acceptability of building and managing wetlands for nonpoint pollution control in a 182-hectare (450-acre) site (Hey 1988). Restoring 10% of the lost wetlands along the Mississippi River (2.5 million ha, or 6.2 million ac) in a 15-year period would cost $24 billion, or $988/ha ($400/ac). The annual operating cost would be $160 million, or $64/ha ($26/ac). Such a restoration effort would require an annual investment of $247 million.

Wengrzynek and Terrell (1990) studied several prototype nutrient/sediment control systems for controlling nonpoint-source pollution from cropland—namely, watershed land treatment practices, sediment basins, grass filter strips, wetlands, deep ponds, and polishing areas. These systems were designed to reduce soluble phosphorus, nitrogen, organic matter, bacteria, and fine sediments in lakes and streams. Construction costs ranged from $14,000 to $22,500 for systems between 8.5 and 66 hectares (21 and 163 acres) in size, or $1,647–$341/ha ($667–$138/ac), respectively. Average annual costs of construction and maintenance were $49/ha ($20/ac).

Prato et al. (1995) evaluated the benefits and costs of converting cropland in two Missouri counties to wetlands. Results showed that conversion was economically feasible when waterfowl hunting benefits were high, a restored wetland was enrolled in the Wetland Reserve Program, and the landowner received full cost sharing on wetland restoration costs. When hunting benefits are low, wetland conversion is not economically feasible. This study suggests that it would be economically feasible for a landowner to convert hydric cropland to a wetland, provided the revenue from waterfowl hunting leases on the wetland exceeds wetland maintenance costs.

CHAPTER 3

Role of Economics in Policy Analysis

Economics can play an important role in identifying least-cost policy strategies that produce the water quality that society desires. An economic framework provides a foundation for coordinating policy formulation among different layers of government, as well as ensuring consistent, fair, and unifying policies across geographic space. Because correcting pollution problems often requires changing the behavior of polluters, it is important to have a conceptual model of that behavior. From an economic perspective, polluters operate within a profit-maximizing economic framework. Thus, one can think of water quality protection policies as altering some of the economic variables a polluter considers when making daily production decisions.

Economics is only one of many factors included when public policy decisions are made. This is certainly true for environmental issues when the public's values have great influence on policy decisions. What follows is a discussion of the economic rationale for our analysis (that also helps explain what drives it), as well as identification of some of the assumptions central to this and other economic analyses.

3.1 GOAL OF ENVIRONMENTAL POLICY

The fundamental goal of environmental policy is to get polluters to treat the external costs of pollution as a cost of production, a process termed "internalizing the costs." This goal can be accomplished by inducing (through economic incentives, such as taxes and subsidies) or by requiring (through standards and regulations) polluters to internalize the external costs that they impose on society through their pollution-related activities.[1] Ideally, the resulting level of pollution control is an efficient solution, or one where the expected net economic benefits to society are maximized. Expected net economic benefits are defined as the private net benefits of production (such as aggregate farm profits) minus the expected economic damage cost of pollution. Note that decisions must be made based on the expectation of damage levels because, when decisions are made, it is impossible to accurately predict damages due to the varying nature of pollutant runoff and transport. Consequently, the efficient solution is often referred to as the *ex ante* efficient solution, meaning that it is the expected outcome as opposed to the actual or realized outcome.

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[1] While we do not discuss this explicitly, existing market distortions that are outside of the regulatory agency's control must be taken into account when designing optimal incentives. Otherwise, the performance of incentives will be limited. A variety of agricultural policies—such as price floors, target prices, and deficiency payments—that are designed to support farm income also have the effect of stimulating production. The resulting use of more chemical inputs and more intensive land use may lead to increases in nonpoint-source pollution (Miranowski 1975; Reichelderfer 1990a; Ribaudo and Shoemaker 1995). The FAIR program has phased out many of these policies, explicitly to reduce market distortions. Other programs, such as acreage retirement programs and paid land diversion, are supply-control programs that may help to offset the effects of some support policies. Recently, some supply-control programs and other agricultural conservation programs (e.g., Sodbuster and Swampbuster) have been targeted to environmentally sensitive land and linked to agricultural support policies.

3.1.1 Efficiency and Benefit–Cost Analysis

The economically efficient solution to pollution problems can be defined by three conditions:

- *For each input and each site, the marginal net private benefits from the use of an input on the site equal the expected marginal external damages due to the use of the input.* The last unit of the input used in production should provide an equal increase in net private benefits and expected damages. This condition is violated, and an economically efficient outcome is not achieved, when farmers ignore the benefit–damage trade-offs associated with most input use. The result is higher (lower) use of pollution-causing (-mitigating) inputs and runoff levels that are above the economically optimal level.
- *An acre of land should be brought into production as long as profits on it are larger than the resulting expected increase in external damages.* Under this condition, the benefits (or profits) from allowing an acre of land into production should exceed the expected social costs of the production activity. This condition defines the optimal amount of land in production. Marginal acreage is defined as land where the profits from production activity equal the activity's expected contribution to damages in the efficient solution (i.e., it is on the margin). Acreage where production activities generate a positive (negative) difference between profits and expected damage contribution is defined as infra-marginal (extra-marginal). From an economic perspective, it is only efficient for the marginal and infra-marginal acreage to be in production.
- *Technology should be adopted on each site such that the incremental impact of that technology on profits (relative to the next-best alternative) is greater than or equal to the incremental impact on expected damages.*

These three efficiency conditions directly address the need to recognize economic trade-offs involving farm profitability and water quality. Together, the conditions imply that farmers must sacrifice some profits to improve water quality if they are currently operating at maximum efficiency and only considering their private costs, all else remaining the same. The challenge is to define an analytical framework that can be used to guide the choice of a policy alternative that will achieve the socially optimal trade-off. Benefit–cost analysis is such a framework.

Benefit–cost analysis is an analytical approach that, in principle, eliminates individual and group biases associated with decision making by heuristics, intuition, or consensus. Given an objective of maximum net economic value or economic efficiency, benefit–cost analysis provides a set of definitions and procedures for measuring benefits and costs and determining optimal policy (Freeman 1994). In doing so, benefit–cost analysis has the potential to rationalize policymaking and ensure the optimal outcomes of policy decisions (Fisher et al. 1986). Promoted as the empirical technique of choice for determining many policy decisions (e.g., U.S. Executive Order (E.O.) 12291, 12866), benefit–cost analysis has a firm foundation in microeconomic theory and management accounting practice, particularly when assessing the net value of a policy or project when the underlying objective is economic efficiency (Dasgupta and Pearce 1972; Mishan 1977; Sassone and Schaffer 1978; Thompson l980; Gramlich l981; Sugden and Williams l985).

Of course, most resource and environmental policy is not based primarily on economic efficiency criteria, either because decision makers have additional objectives (e.g., equity considerations, intergenerational effects, social risk aversion) or because the information base required to define all the benefits and costs cannot be obtained. Thus, benefit–cost analysis might be best thought of as a set of procedures to help organize the available information, rather than a straightforward set of decision rules (Freeman 1994). While this perspective on benefit–cost analysis does not attempt to define the ultimately rational policy choice, it is capable of meeting the basic requirements of E.O. 12866: analyses that are economically sound, are based on appropriate data and methods, are correctly interpreted, identify all affected parties, and estimate, where possible, all relevant costs and benefits (Schaub 1997). The level of sophistication and method in benefit–cost analysis can range from simple comparisons of directly and readily measurable financial factors to multifaceted techniques that incorporate tangible and intangible factors (Clarke and Stevens 1997). In general, all benefit–cost analyses attempt to identify and measure the benefits and costs attributable to a policy and to compute the policy's net value (OMB 1996).

3.1.2 Cost-Effectiveness Analysis

Although pure Pareto efficiency and decision making based on benefit–cost analysis comprise the conceptually ideal approach for addressing pollution problems, efficiency is not generally attainable in practice if the damage and/or pollution-transport functions are unknown or poorly understood, as is generally the case with nonpoint-source pollution. While these problems do not prevent the design of economically sound pollution control policies, they do require that policies be based on alternative objectives. Baumol and Oates (1988) suggested designing pollution control policy to meet an emission or ambient pollution target when damages are unknown. For example, without information on damages, the regulatory agency's goal when designing policy would be to attain a mean ambient water quality goal at least cost; alternatively, without information on pollution transport to water resources, its goal would be to attain specific mean runoff goals at least cost.

Cost-effective solutions are not Pareto-efficient because water quality damages—and, thus, the benefits from reducing pollution—are not a consideration (since they are unknown) (Shortle 1987, 1990). As a result, the traditional benefit–cost analysis is truncated in that benefits of proposed policy actions may not be measurable, given available information. In these cases, a cost-effectiveness analysis is designed to provide information about the economic trade-offs associated with different types of management strategies. Cost-effectiveness analysis is considered appropriate whenever it is impractical or impossible to consider the monetary value of the benefits provided by alternative policies (OMB 1992). Under a cost-effectiveness analysis, a policy can be considered cost-effective if, on the basis of life-cycle cost analysis of competing alternatives, it is determined to have the lowest costs for a given amount of benefit, however benefit is defined.

3.1.3 Second-Best Policies

The potential complexity of efficient or cost-effective nonpoint source pollution policies can make their administration and implementation difficult and costly. At a minimum, the regulatory agency would need perfect information about the production and runoff functions for each site. Some optimal policies are site-specific and require the regulatory agency to perfectly monitor technology and input usage on each site, including those inputs that are not sold in the market.

The costs associated with obtaining the necessary information for determining and applying site-specific policies and monitoring input usage and technology choices can be substantial. These costs are relevant and should not be ignored when a potential policy is designed or analyzed. Under these conditions, the costs of obtaining an efficient or cost-effective outcome should be weighed against the decreased benefits that may result from taking a more uniform, but informationally less intensive, approach to policy design. Policies that are specifically designed to reduce information and administrative costs at the expense of efficiency or cost-effectiveness are referred to as second-best policies.[2] Most, if not all, nonpoint-source pollution policies can be considered second-best polices.

3.2 POLICY-RELEVANT CHARACTERISTICS OF NONPOINT-SOURCE POLLUTION

The defining characteristics of nonpoint-source pollution, which are described in this section, are important because they will influence the performance of various pollution control options.

3.2.1 Observability of Runoff and Loadings

Some important aspects of agricultural nonpoint-source pollution are difficult to measure, as in the inability of regulators (and farmers) to observe runoff from a field and loadings into water systems. Also, monitoring the movement of nonpoint-source pollutants is impractical or prohibitively expensive. Impacts on ambient water quality can be observed, but because nonpoint pollution is generated over the land, enters water systems over a broad front, and has substantial natural variability, it is generally impossible to use ambient quality measures to make inferences about where pollutants enter the water and from which cropland the pollutants originate.

The inability to observe loadings would not be such an obstacle if there were strong correlation between ambient quality and some observable aspect of the production process. For example, the quality of a shallow aquifer that is entirely overlain by cropland is directly related to how the fields are managed. A policy could then be directed at the production process with a reasonable expectation of the water quality impacts. However, such correlations do not often exist, and where relationships can be established, they are unlikely to hold up across a range of conditions. Because regulators cannot infer producers' actions by observing the state of water quality, they are uncertain as to whether poor water quality is due to nonpoint sources of pollution failing to take appropriate actions or to undesirable states of nature (e.g., high rainfall) (Malik et al. 1992).

Finally, observations on the use of production inputs, which are critical for predicting or forming expectations about nonpoint-source pollution, may also be unobservable or prohibitively expensive to monitor. For example, there is a close correlation between the chemical contamination of ground water and the amount of a chemical applied and soil type. The chemical characteristics of the pesticide, soil characteristics, and soil depth to ground water can all be determined. However, chemical application rates and timing are generally not observable to a regulating agency without costly and intrusive monitoring. Producers have a special knowledge about their operations that they may not willingly share with potential regulators.

[2]Efficient policies are first-best. Cost-effective policies are also technically second-best because they are inefficient. For simplicity and consistency, we distinguish between cost-effective and alternative second-best policies.

3.2.2 Natural Variability and Pollution Flows

The nonpoint-source pollution process is influenced by natural variability due to weather-related events (e.g., wind, rainfall, and temperature). As a result, a particular policy may produce a distribution of water quality outcomes, rather than a single outcome (Braden and Segerson 1993). This by itself does not prevent attainment of *ex ante* efficiency through the use of standard policies. However, it greatly complicates policy design. For example, nearly all soil erosion occurs during extremely heavy rain events. Practices that control erosion from "average" rainfalls but fail under heavy rain events will likely be ineffective in protecting water resources from sediment. In addition, natural variability may limit the effectiveness of models in predicting relationships between production decisions and water quality, especially if the models are based solely on knowledge of production decisions.

Natural variability has important implications for cost-effective policies that attempt to achieve ambient or runoff targets at least cost. The natural variability of the nonpoint pollution process limits policies from being able to attain specific targets. Instead, as mentioned above, policies produce a distribution of results. Therefore, runoff and ambient targets must be specified, along with the reliability with which that goal is to be achieved (Shortle 1987, 1990). For example, the goal can be defined in terms of mean pollution levels. Alternatively, a nutrient control policy may require that an ambient goal of 10 mg/l be met for 75% of the samples taken during a year.

3.2.3 Heterogeneous Geographic Impacts

The characteristics of nonpoint-source pollution vary over geographic space due to the great variety of farming practices, land forms, climate, and hydrologic characteristics found across even relatively small areas. This site-specific nature of nonpoint pollution has important policy implications. For example, even if models could be developed to measure runoff and loadings, they would have to be calibrated for the site-specific qualities of each individual field. The information required for such calibration would be significant and possibly unavailable. Therefore, consideration of the spatial characteristics of cropland, pollution transport, and dispersion of pollutants introduces additional uncertainties into the estimation of loadings into water resources (Miltz et al. 1988). Because of these difficulties, flexible policy tools that can provide optimal pollution control under a variety of conditions would have advantages over tools that are not self-adjusting (Braden and Segerson 1993).

3.2.4 Transboundary Effects

The effects of agricultural nonpoint-source pollution can often be felt far from their source. Chemicals with long half-lives and sediment (conservative pollutants that tend to maintain their properties in a water environment) can affect water users far from where they originate. For example, much of the herbicide atrazine and nitrates that enter the Gulf of Mexico each year via the Mississippi River are applied to cropland in the upper Corn Belt states of Minnesota, Iowa, and Illinois (Goolsby et al. 1995).

3.2.5 Uncertain Water Quality Damages

As with most types of pollution, the economic damages associated with water quality impairment are often difficult to observe or ascertain. Knowledge of the relationship between economic damages and water pollution is essential for establishing water quality goals or incentive levels that maximize societal welfare. Adding to the difficulty is the fact that the impacts of pollution on water quality are often nonmarket impacts. For example, the nitrification of Chesapeake Bay is believed to reduce the bay's submerged aquatic vegetation levels. Though there is no market for submerged aquatic vegetation, it has economic value because it provides habitat for economically valuable fish populations, among other things.

Without organized markets, information on the value of water quality may be difficult to obtain. Even if these impacts are observed and can be attributed to specific sources, valuation requires the use of a nonmarket valuation technique, such as travel cost or contingent valuation (Ribaudo and Hellerstein 1992). However, such exercises are both time-consuming and costly, and their reliability is questionable.

3.2.6 Time Lags

The movement of a pollutant off a field to the point in a water system where it imparts costs on water users may take a considerable amount of time. Time lags of this sort have two policy implications. First, observed ambient water quality conditions may be the result of past management practices, or of polluters who are not longer in operation. Second, the results of a policy may not be immediately apparent, making it difficult to assess its actual effectiveness.

3.3 ECONOMIC CONSIDERATIONS IN SELECTING POLICY INSTRUMENTS

Policy instruments for controlling water pollution that have been considered or tried at the federal, state, or local level fall into five general classes: economic incentives, regulation, education, liability, and research and development. Policymakers must consider a number of important economic, distributional, environmental, and political characteristics when selecting one of these instruments.

3.3.1 Economic Performance

The instruments differ in their ability to maximize net social benefits by correcting an externality. Some may only be able to achieve a second-best solution because external pollution costs are not fully accounted for when production decisions are made. The policy instruments also distribute costs of pollution control differently among polluters and between polluters and the rest of society. For example, subsidies place the burden of pollution control on taxpayers, while taxes place the burden on polluters.

3.3.2 Administration and Enforcement Costs

The costs of administering and enforcing a water quality protection policy are related to a variety of factors, including the nature of the pollution problem, the legal system, and the information required to implement an instrument efficiently. These costs have particular importance for policies aimed at controlling nonpoint-source pollution.

Nonpoint runoff is difficult to monitor due to its stochastic and diffuse nature. Likewise, ambient concentration measurements and chemical loss estimates may be subject to error. In addition, while it is straightforward to monitor the use of purchased inputs, it may be difficult to monitor the use of all polluting inputs. If the costs of detecting noncompliance are too high, polluters will be able to avoid compliance, and the effectiveness of the policy will be degraded (Braden and Segerson 1993). Thus, the policy’s administration and enforcement costs need to be weighed against its potential environmental benefits.

3.3.3 Flexibility

Flexibility refers to a policy instrument's ability to provide effective control in the face of changing economic conditions (e.g., changes in input and output prices or the availability or new technologies), changing environmental conditions (due to the stochastic and highly variable nature of nonpoint pollution), and heterogeneous physical conditions (due to the site-specific nature of nonpoint pollution). To the extent that agricultural nonpoint pollution and resulting water quality impacts are a function of these changes, policy tools that can adjust without regulator intervention will be more efficient over the long run than those that require regulators to make the adjustments.

3.3.4 Innovation

A desirable characteristic of a policy instrument would be its ability to encourage and reward farmers for using their unique knowledge of the resource base to find better ways of meeting policy goals (Shortle and Abler l994; Bohm and Russell 1985; Braden and Segerson l993). Instruments that provide these incentives are more likely to achieve cost-effective control than those that do not.

3.3.5 Political and Legal Feasibility

Even though a number of policy instruments are capable of achieving an economically efficient outcome, they may not be perceived to be equal for legal or political reasons. The difficulty in observing nonpoint runoff may be a source of legal problems for instruments using runoff or ambient quality as a base. For example, it may be difficult to hold individual farmers legally responsible for observed water quality damages when the sources of nonpoint pollution cannot be observed. The stochastic nature of nonpoint pollution also makes it difficult to accurately infer damages or runoff based on farm practices (Shortle 1984; Tomasi et al. 1994). In addition, ambient pollution levels may be the result of past management decisions due to time lags involved with the pollution transport process. Thus, some contributors to the ambient pollution level may no longer be in operation, possibly leaving current farming operations to unfairly bear the burden of remediation.

An instrument's political feasibility also may be related to ethical and philosophical arguments. For example, farmers may believe that their right to farm gives them the right to pollute (within reasonable limits). Taxes and permits may be politically unpopular among farmers because they shift pollution rights from farmers to the victims of pollution. Alternatively, a subsidy shifts pollution rights to the farmers. This position may be protested by the victims of pollution and by industries that are legally required to reduce pollution. In an era of widespread anti-tax sentiment, a tax-based environmental policy may be impossible to implement, despite desirable efficiency characteristics.

3.3.6 Implementation Basis

The basis on which a policy instrument is implemented has a bearing on its performance. The basis refers to the point in the pollution stream to which the instrument is applied. Instruments can be applied either to farmers' actions or to the results of their actions. For point sources, the preferred basis is discharge because it is directly related to water quality and is easy to observe (Baumol and Oates 1988). However, the choice is not so clear for nonpoint sources due to the difficulties associated with monitoring and controlling runoff from cropland and loadings to water resources.

Potential bases that have been proposed for nonpoint pollution include ambient pollution levels, expected runoff levels, input use, technology, and output. Policies that control variables most closely correlated to water quality are preferred to those that are more indirectly related (Braden and Segerson 1993). Runoff and ambient pollution concentrations are two bases that are closely related to water quality. Policy bases, such as output, are not likely to be highly correlated with water quality. Directing policy instruments at bases that are only indirectly correlated with water quality may lead to unrelated effects and inefficient management.

3.4 POLICY INSTRUMENTS FOR ACHIEVING GOALS

Design-based policies are based on observable aspects of production, such as variable input use or choices of production technologies. Such policies can achieve effective control if they include all factors of production that can affect water quality and if they take into account local agronomic and hydrologic conditions (Ribaudo et al., in press). The costs of doing so, however, are often prohibitive. Alternatively, second-best, design-based instruments could be applied to a limited (truncated) set of inputs and/or technologies, and applied uniformly within a region. They could also be designed with limited information on the part of the regulatory agency to help control administration costs. Such instruments may be effective in controlling nonpoint pollution if the inputs/technologies chosen as bases are highly correlated with water quality.

For a given instrument base, economic incentives (taxes and subsidies) or standards can be used to achieve identical policy goals. However, use of each instrument type will most likely have different distributional consequences for farm profitability. Distributional disparities will be greater the greater the heterogeneity of land, the more uniformly instruments are applied across a region, and the more uncertainty the regulatory agency has about farm-specific information when designing policies. In general, incentives provide more flexibility than standards because farmers are free to adjust their production practices to take advantage of personal knowledge and to react to changing market conditions. Standards provide more certain control when uncertainties in the relationships between production and water quality are high.

Incentives and standards will also have different administrative characteristics. The information required by the regulatory agency in setting design-based standards and incentives is very similar. However, monitoring may be easier for incentives that can be applied through existing markets. For example, a uniform fertilizer tax can be implemented as a sales tax, whereas a fertilizer standard requires that each farm be monitored for fertilizer use. Taxes also have the additional advantage of generating revenue that could be used for supporting the administration of the water quality policy; for funding supporting programs, such as education and research; or for retiring marginal land. For example, the sales tax on fertilizer in Iowa is currently being used to support the state's nutrient management programs. While the tax rate is currently too low to be called an environmental tax, research and education efforts may be increasing the efficiency of fertilizer use. That the nitrogen fertilizer application rate on corn is much lower in Iowa than for the other Corn Belt states is circumstantial evidence that research and education are having an effect (USDA 1996).

It is not possible to make a general statement about the relative performance of incentives and standards in a world with asymmetric information and second-best policies. There are situations in which each is preferred. Shortle and Dunn (1986) compared the use of input standards and input incentives applied to a single farm and designed to achieve an efficient solution when asymmetric information exists. Ignoring transaction costs, they found that appropriately specified input incentives should generally outperform input standards and expected runoff incentives and standards, given the characteristics of nonpoint-source pollution and the information typically available to a regulatory agency. These results, however, do not necessarily carry over to the case of multiple farms and/or second-best policies, where administration costs are considered. Weitzman (1974) examined price and quantity policies under asymmetric information and showed that, where the marginal cost curve is nearly flat, an error in setting a tax could result in large deviations from the desired result, making standards the preferred instrument. Alternatively, when the benefit function is closer to being linear, price-based policies are superior. Stavins (1996) showed the choice to be more complex when the uncertainty associated with the benefits and costs of pollution control is correlated. A similar analysis applies when uniform policies are used when heterogeneities exist. Generally, each situation must be assessed individually.

Helfand and House (1995), found uniform input taxes to result in lower welfare costs relative to input standards when attempting to meet a desired water quality goal. These results held for taxes and standards applied to all inputs contributing to pollution, and also for the case of a truncated input base. Lichtenberg (1992) found that standards may be preferable to incentives when a specific input-reduction goal is desired. For example, a standard would be preferred in a situation where a particular chemical is clearly detrimental to water quality and application rates need to be limited, or the chemical banned from use. Setting a tax to optimally meet an input-reduction goal requires knowledge of the farm-specific demand for that input. Such information is not likely to be available to a regulatory agency. Design standards, in the form of input use limits, would be much easier to implement in this case, even though the distributive properties might be poor. Other examples where design standards might be preferred include chemigation (using irrigation equipment to apply chemicals along with water), chemical use on sandy soils, the use of vegetative buffers, and animal waste storage and use.

One conclusion from the above review would be that the differences in agriculture and hydrology across regions probably do not favor a single policy tool. Multiple instruments have a role when a single instrument is inefficient because of the characteristics of nonpoint-source pollution (Braden and Segerson 1993). In his study of price- and quantity-based policies, Weitzman (1974) concluded that mixed price/quantity policies may give the best results in some situations, depending on the characteristics of the polluters and receiving waters. In a review of pollution policy tools, Baumol and Oates (1979) conclude that "effective policy requires a wide array of tools and a willingness to use each of them as it is required."

Abler and Shortle (1991) reviewed the merits of a variety of tools (including education, design standards, performance standards, input taxes, input subsidies, performance taxes, and research and development) for reducing agricultural nonpoint-source pollution. Using a set of evaluation criteria based on both economic and administrative attributes, they could not identify a single dominant tool. Each had its strengths and weaknesses. Which tools are actually preferred in a particular setting depends on the weights applied to the various attributes.

Shortle and Abler (1994) evaluated a mixed scheme consisting of marketable permits for polluting inputs, combined with a tax on excess input use and a subsidy for returned permits. Such a scheme can be implemented without the use of information on farm profits or off-site damage costs. This approach was generally shown to be preferred to policies based solely on design incentives. Optimal implementation could still entail large administrative costs, but the structure should offer opportunities for increased efficiency over input-based tax and license schemes that have been suggested as potential policies.

3.5 SCOPE AND OUTLINE OF THE ECONOMIC ANALYSIS

Under Executive Order (E.O.) 12866, agencies determine whether a potential rule or regulatory action is "significant" and therefore subject to the requirements of the E.O., which include drafting an economic analysis and review by the Office of Management and Budget (OMB). While the Gulf of Mexico Hypoxia Assessment is not itself a rule-making or regulatory activity, analysis of the economic impacts of hypoxia and potential actions for its mitigation in the Gulf would almost certainly involve some form of federal intervention.

Federal intervention could take many forms and would most likely interact with current state and federal ambient water quality criteria and designated uses for inland surface waters, enclosed bays, and estuaries. Thus, the economic analysis of hypoxia in the Gulf of Mexico was designed to be relatively broad in scope, seeking to identify major benefits and costs by using available information and numerical models.

Given the scope of this assessment, the economic analysis requires assumptions about the source of benefits and costs associated with hypoxia and hypoxia mitigation, as well as how mitigation activities might be implemented. The assumptions used in the analysis were based on standard practice within the economics profession, information provided by other groups participating in the hypoxia assessment, and past experience with mitigation activities associated with large watersheds. To account for some of the uncertainty in these assumptions, we estimated a range of scenarios that may be used to bracket the likely effects of hypoxia and hypoxia mitigation activities. In addition, we attempted to qualitatively identify potential benefits and costs that could not be estimated due to a lack of information or relevant research, but that may be useful in directing future research associated with hypoxia.

3.5.1 A Cost-Effectiveness Economic Analysis

An immediate difficulty with conducting an economic analysis of Gulf of Mexico hypoxia concerns the lack of measurable benefits (and thus the benefits side of a benefit–cost analysis calculation) that may accrue from reduced hypoxia. The Topic 2 report examined historical time-series and geographical data to discern the effects of hypoxia on the Gulf's ecosystem and fishing industry over the last 30 years. This assessment suggested that given the available data, no effect in the shrimp, snapper, or menhaden fisheries data could be attributed with high confidence to hypoxia. They also noted that the failure to identify hypoxic effects in the fisheries data was consistent with the results of the broader ecological study conducted by the group.

It is worth emphasizing, as did the Topic 2 report, that failure to identify hypoxic effects in the commercial fisheries data does not mean that they do not exist. But, if hypoxic effects do exist, their magnitude must be small in relation to other sources of variability in the data. Another constraining factor was that the data needed to identify indirect and nonmarket effects of hypoxia, such as its impact on tourism and recreation, were not readily available for the Gulf. In addition, the inability to identify hypoxic effects in the historical data does not imply that larger effects would not occur should hypoxia continue or expand in the Gulf. In fact, experience in other geographic areas indicates that the effects of hypoxia become progressively greater as the frequency and extent of hypoxic events expand (Caddy 1993).

Nonetheless, given the lack of measurable economic benefits from reduced hypoxia, the economic analysis undertaken below was restricted to a cost-effectiveness analysis that sought to identify least-cost policies for attaining a representative reduction in nonpoint nitrogen runoff to the Gulf. The fixed target level of a 20% reduction in nitrogen loss was suggested by the Topic 5 team as a reasonable level that could be attained given current technology and that could decrease the incidence of hypoxia in the Gulf.

3.5.2 Criteria for Choice of Reductions

Efficacy was an important criterion used by the Topic 5 team when considering different actions to reduce nitrogen loss. While the goal of the economic analysis was to evaluate and compare the costs and benefits of such reductions to the Gulf, benefits could also accrue within the Mississippi River Basin. Although we are unable to fully document and account for these potential benefits, we have attempted to include them in this discussion as examples of unmeasured or unmeasurable factors. Of course, a limitation on the extent of potential in-basin benefits is that phosphorus—not nitrogen—tends to be the limiting factor in freshwater ecosystems. Thus, very little research has focused on identifying benefits from nitrogen reduction in freshwater systems.

In conducting the economic analysis, we attempted as much as possible to follow the technological guidelines provided in the Topic 5 report. Where we had existing analytical capacity, we simulated a wide range of potential policy options concerning nutrient reduction. This allowed us to look at impacts from different reduction levels from a particular action or set of actions. Our concern was finding the specific range for the various potential reductions that did not result in extremely high social or economic costs. While this exercise was independent of the recommendations of the Topic 5 report, the implications of our analysis tended to converge with those recommendations primarily on technical considerations of efficacy.

3.5.3 Background and General Assumptions

The Mississippi River Basin encompasses the bulk of the agricultural land area devoted to basic agricultural commodities. A comparison of the crop acreage within the basin and across the entire U.S. reveals that the land in the basin comprises the bulk of quality rain-fed land east of the critical 100-degree meridian, or 22-inch rainfall line. As the maps from the Topic 3 report show, it is also the area of highest nutrient use across broad sections of the landscape.

Each specific policy simulation conducted as part of the economic analysis has a specific set of assumptions, which are detailed with the analysis. In addition, the analysis used the following important general assumptions:

- The current “reformed” agricultural policy will continue, where producers are free to make choices about the crops they grow and where acres are not taken out of production just to regulate supply.

- The analytical model upon which the policy simulations are based assumes profit-maximizing behavior by the producers. This assumption also implicitly, if not explicitly, conditions our analysis of the simulation results and our judgments concerning producer or individual firm reactions to policy options.
- The Mississippi River will continue to be a critical transportation corridor, and the basin will have a stable hydrology. Dredging and lock and dam maintenance will continue, constraining (to some extent) the potential policy options.
- From nonpoint sources, such as soils under fertilization, there can be substantial time lags before actions like reductions in nitrogen fertilization are reflected in reduced nitrogen loss from the soil. This is a critically important and uncertain factor when assessing the efficacy of a practice or activity in reducing nitrogen losses to the river system. We also recognized that the effects of changing policies or placing new activities or practices in operation are likely to be measured in years rather than months.
- The baseline used for investigating potential benefits in the Gulf is detailed in the Topic 2 report, but was essentially the status quo in the Gulf's commercial fishing industry. The numerical baseline used for empirically estimating the costs of nutrient-reduction strategies to agricultural industries in the Mississippi Basin was described in this chapter. The U.S. Mathematical Programming (USMP) model served as the baseline for estimating both the costs of the nutrient-reduction strategies and the benefits to water resources within the basin (but not in the Gulf).

CHAPTER 4

Costs of Alternative Control Options

The most tractable strategies for reducing nitrogen runoff to the Mississippi River system involve altering the use of nitrogen fertilizer inputs (which can be observed), or enhancing the landscape's ability to filter nitrogen that "leaks" from cropland. In this chapter we assess both the costs of alternative fertilizer management strategies and the strategies for expanding the acreage of restored wetlands and riparian buffers and thereby increase nitrogen filtration.

If hypoxia reduction is attempted by policy initiatives that decrease nutrient loadings to the Gulf of Mexico, then substantial nutrient management costs are likely to be borne by agricultural industries in the Mississippi Basin. Assuming constant levels of efficiency, decreased nutrient use must lead to reduced agricultural productivity in the basin and to reduced total production activity as marginally fertile land is retired. If increased nutrient management efficiency is assumed, then the adoption of alternative technologies may raise the total cost of production and lower the basin's total output. These costs are both direct (associated with forcibly removing low-productivity acreage from production) and indirect (in terms of the economic impacts on the supply sector generated by reduced demand for purchased agricultural inputs). In either case, production activity could shift to areas outside the basin, and with the production activity would go the associated nutrient use. This increased nutrient use in the rest of the U.S. could generate costs similar to the costs of hypoxia, although the exact nature of the impact will depend on the ecological structure of the receiving waters. Increased agricultural product prices should also be considered a potential cost, borne by consumers throughout the U.S., as long as the land retired in the Mississippi Basin is not compensated for by increased production on remaining land in the basin or the rest of the U.S. Of course, increasing product prices imply that some of the nutrient management and land retirement costs to producers may be offset by increasing gross revenues.

We simulated the sector's response to these alternative management strategies with the U.S. Mathematical Programming (USMP) model of the nation's agricultural sector (described in the following section). The analysis of alternative fertilizer management strategies relies solely on USMP; the analysis of the various filtering options uses both a screening procedure to identify the acreage to be restored to wetlands or buffers, and USMP to analyze the economic impacts on the agricultural sector of removing this land from production. Environmental impacts for field-level changes—such as changes in rotations, tillages, and fertilizer applications or retirement from cropping—are estimated in USMP based on EPIC (Erosion Productivity Impact Calculator) simulations. (See Appendix C for further discussion of EPIC.) Nitrogen reductions from intercepting runoff with restored wetlands or riparian buffers are estimated outside USMP based on values from the literature identified in Chapter 5 of this report.

4.1 USMP MODEL FOR AGRICULTURE

The USMP model was developed by USDA's Economic Research Service to analyze the effects of government commodity programs and environmental policies on the U.S. agricultural sector and the environment. It captures the effects of these policies on commodity prices and quantities, net returns to producers, net social benefit, and environmental emissions. It covers 10 crops (corn, sorghum, oats, barley, wheat, rice, cotton, soybeans, hay, and silage) and some 16 live animal products, the principal being associated with dairy, swine, poultry, and beef cattle. Several dozen processed and retail products are also represented, including dairy products, pork, beef, soybean meal and oil, and livestock feeds. Markets represented include domestic use (food, industrial uses, and livestock feed), trade (import and export), and inventory (commercial and government). The model explicitly recognizes that these effects will most likely vary significantly across the country, and that this variation needs to be taken into account, especially with respect to analysis of environmental issues, by dividing the U.S. into 45 production regions. The boundaries of these regions are defined by the intersection of the 10 USDA farm production regions and USDA land resource regions.

A key feature of USMP that makes it well suited for this analysis is that it explicitly represents the various management practices used by farmers to produce crops in each production region. These practices consist of crop rotations, which can include up to four crops (including hay and fallow); tillage practices (conventional, conventional with moldboard plow, mulch, ridge till, and no till); and reductions in fertilizer application (Table 4.1). Fertilizer is applied in single or split applications and at rates varying continuously up to 60% from baseline levels, depending on the predominant practices in a region. Overall, nearly 2,400 individual production technologies representing distinct management alternatives are represented in the model. Thus, crop production can adjust to restrictions placed on nitrogen or land use by altering any or all of the following: acreage planted, crop mix, rotations used, tillage practices, and fertilizer application rates.

The management practices and the acreage devoted to them were identified from information contained in the 1992 National Resources Inventory (USDA 1995a), the Cropping Practices Survey (USDA 1995b), and the Farm Costs and Returns Survey (USDA 1995b). These USDA surveys were all designed to be statistically representative of their target populations. As a result, the set of production technologies used in USMP is representative of current management practices.

USMP also contains a set of associated environmental emission indicators for each of the management practices represented in the model. This includes soil erosion (both water and wind), nitrogen loss (leaching, runoff, and atmospheric), and pesticide loss, as well as indicators of soil depreciation and greenhouse gases.

USMP is a comparative, static price, endogenous, spatial equilibrium model based on an extension of the methodology found in McCarl and Spreen (1980). It uses a set of nested non-linear allocation functions to represent the substitution possibilities among production techniques. These functions derived from those relationships, both technical and economic, are derived from neoclassical theory. As a result, production practices can adjust to incentives based on changes to their net returns without placing bounds or flexibility constraints on their use.

TABLE 4.1. Illustrative cropping systems in the USMP model, by USDA farm production region.[1]

CORN BELT	**Continuous Corn (lbs/acre)** Convent'l.	Convent'l. MB[2] Plow	No till	**Corn–Soybean (lbs/acre)** Convent'l.	Convent'l. MB[2] Plow	No till
Total nitrogen applied[3]	**143**	**147**	**118**	**70**	**67**	**67**
Soil erosion—water	5	7	4	5	8	4
Nitrogen loss in solution	9	9	6	8	6	8
Nitrogen loss in sediment	30	38	31	34	44	35
Nitrogen leaching	1	2	7	1	1	1
Denitrification	17	18	9	12	14	11
Total nitrogen lost	**57**	**67**	**46**	**55**	**66**	**56**

NORTHERN PLAINS	**Continuous Corn** Dry Land Convent'l.	Dry Land No till	Irrigated Convent'l.	Irrigated No till	**Wheat–Fallow** Convent'l.	Convent'l. MB[2] Plow
Total nitrogen applied[3]	**68**	**80**	**172**	**184**	**14**	**14**
Soil erosion–water	3	3	2	2	3	2
Nitrogen loss in solution	7	8	24	32	4	4
Nitrogen loss in sediment	16	17	20	18	11	7
Nitrogen leaching	4	7	1	1		
Denitrification	10	13	8	8	4	4
Total nitrogen lost	**37**	**44**	**53**	**58**	**18**	**14**

DELTA	Continuous **Cotton** *Conventional*	**Continuous Rice** *Conventional*	**Continuous Soybeans** *Conventional*	**Continuous Beans & Rice** *Conventional*
Total nitrogen applied[3]	**96**	**125**	**0**	**69**
Soil erosion–water	9	2	7	6
Nitrogen loss in solution	39	26	26	31
Nitrogen loss in sediment	10	2	10	7
Nitrogen leaching	12	12	8	12
Denitrification	3	3	2	3
Total nitrogen lost	**64**	**43**	**46**	**53**

[1]Cropping systems are specific to each of 45 USMP-producing regions (farm production region and land resource region combinations) and representative soils, totaling nearly 2,400 cropping activities.
[2]MB = moldboard.
[3]Nitrogen applications can vary continuously in reductions of up to 60% from baseline levels for each cropping system, and can include split applications on corn.
Source: U.S. Mathematical Programming (USMP) model analysis, Economic Research Service, USDA.

The base or reference projections used have been developed from the 1997 USDA Economic Baseline, the Cropping Practices Survey, and the 1992 National Resource Inventory. For each scenario the model tallies—in addition to changes in supply, use, prices, and farm income—changes in acreage allocated under various rotation and tillage systems and associated environmental indicators for the nation, the USDA's 10 farm production regions, and 45 sub- or land resource regions. See Figure 4.1 for a map of the regions.

The environmental indicators are estimated using the EPIC biophysical model. EPIC uses information on soils, weather, and management practices, including specific fertilizer rates and produces information on crop yields, erosion, and chemical losses to the environment. Management practices and initial fertilizer application rates were set consistent with agronomic practices for the 45 regions as reported in the CPS. Yields and environmental indicators were estimated by running each of the production activities through EPIC for up to 60 years. This process was repeated for nitrogen application rates representing 10%, 20%, 30%, and 40% reductions from their initial values. The results from the EPIC simulations were used to construct four sets of budgets and environmental indicators for each of the initial cropping systems, increasing the total number of crop production activities represented in the model to about 2,400.

Convexity constraints are used to approximate yield response to nitrogen fertilizer for each cropping system. These constraints permit convex combinations of the specified nitrogen application rates to be formed, thereby allowing fertilizer application rates and associated environmental indicators to span the entire range of fertilizer rates simulated.

In addition to the convexity constraints the selection of fertilizer application rates is affected by a risk premium charged to the reduced nitrogen fertilizer application activities. This is based on the notion that farmers, given their uncertainty about growing conditions in any year, are behaving rationally when they apply a constant amount of fertilizer year in and year out based on a yield target. Thus, producers are assumed to operate at that portion of the typical fertilizer response curve corresponding to the upper "shoulder" or inflection representing maximum physical yield at that level of fertilizer application. In particular, these assumptions regarding fertilizer application rates imply that any reduction in fertilizer application from the baseline will result in reduced physical yield, but not necessarily reduced economic returns. However, this type of behavior when evaluated in a partial budgeting framework indicates that farmers could increase their returns by reducing the fertilizer application rate.

Thus, the risk premium being charged represents the uncertainty cost borne by farmers. It reflects the cost they pay to avoid the censoring of the upper tail of the yield distribution, which would occur if they reduced fertilizer application rates. This implies that the only way to get farmers to lower their fertilizer application rates is to provide them with an incentive greater than the risk charge or to introduce technologies that will reduce their uncertainty about growing conditions. (Another approach being offered on a pilot basis is providing insurance against production losses due to low nitrogen levels.)

The charge used on each system is based on the difference between the net returns realized by reducing the fertilizer application rate and that achieved when using the observed application rate. This difference is doubled, based on the assumption farmers will need more than this difference to reduce application rates. This calculation is repeated for each successive reduction in the fertilizer rate until the increase in net return calculated is less than the net return of the application rate before it. At this point, the charge is held constant for the remaining reductions in application rates.

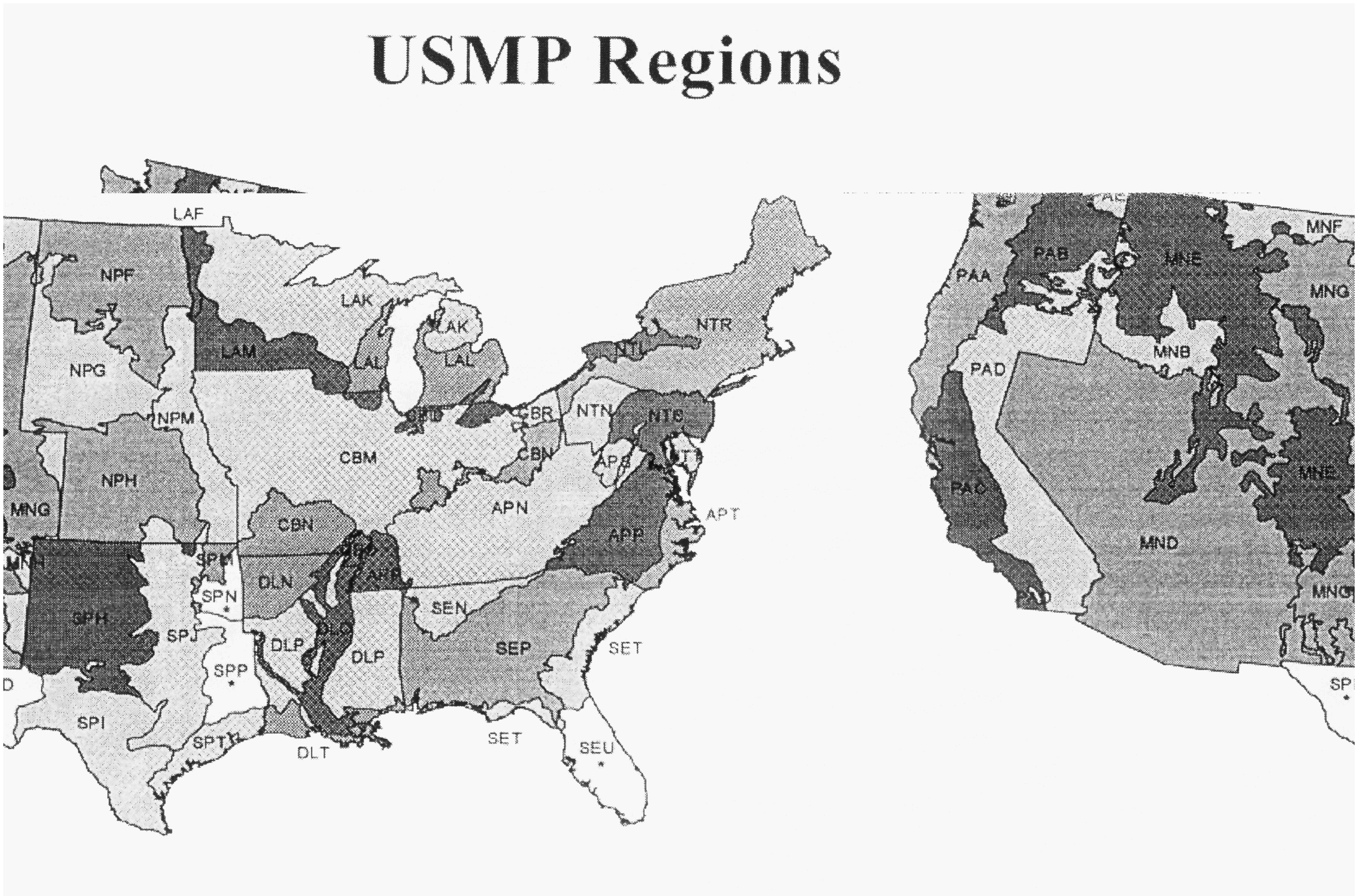

FIGURE 4.1. U.S. mathematical programming (USMP) model regions.
(Data not available for SPD, SPP, SPN, and SEU.)

These assumptions result in overall fertilizer application consistent with observed data reported in Table 1.1 of the Topic 5 report and by Goolsby et al. (in press). The USMP baseline has 6.859 million metric tons of N fertilizer used on crops in the basin, compared with 6.578 million metric tons reported in Table 1.1. USMP N application rates compare closely to application rates from the ERS Cropping Practices Survey.

For the analysis of wetland restoration potential, a hybrid modeling technique was employed, combining (1) a screening procedure to identify acreage and production affected by wetland restoration with (2) an impact analysis conducted using the USMP model. Cropland suitable for restoration to wetland was screened based on profitability, similar to methods reported in Heimlich (1994) and Claassen et al. (1996). All cropland on wetland (hydric) soils in the National Resources Inventory, except artificial wetlands and wetland converted in violation of Swampbuster, is assumed to be eligible for enrollment in an expanded wetland restoration program (WRP). However, a restriction that no more than 25% of total cropland in a local area (8-digit hydrologic unit) be enrolled was assumed. This is similar to restrictions in the Conservation Reserve Program and is intended to minimize impacts on local farm economies.

The costs of permanent easements are assumed to compensate landowners for the opportunity cost of agricultural production on the restored wetland and for all restoration expenses. Easement cost equals full opportunity costs. Opportunity costs are the maximum of the value of crop production, pasture, bottomland hardwood forestry, or pine (drained) forestry. A discount rate of 6% is assumed, and since easements are permanent, an infinite time horizon is used. Landowners are assumed to be indifferent to enrolling land in the expanded WRP or continuing crop or other agricultural production at these rates of compensation.

Crop opportunity cost calculations depend on site-specific crop yields derived from five-year average yields in the county, adjusted for a productivity index applied to soil characteristics. Only variable production costs are considered, since WRP enrollment is assumed to be a marginal change in farm acreage that will not affect fixed costs. No farm program payments are reduced by enrollment in this program. Crop prices assumed are from year 2001 of the February 1997 USDA baseline, and do not account for the reduced prices currently prevailing. Restoration costs are differentiated by drainage condition and region (Table 4.2). Administrative costs, including appraisal, survey, recording, and title fees, are estimated to be 3% of easement costs (Misso 1998).

TABLE 4.2 Cost per acre of land restored.

Region	Fully Drained	Cropped Wetland
Prairie Pothole	100	50
Delta and Southeast	800	600
All Other	500	300

Source: Heimlich et al. 1989.

For the purposes of this study we divided the 45 USMP regions into two groups: those inside and those outside the Mississippi Basin. Because the USMP regions do not follow watershed boundaries, the allocation is fairly crude. However, the most important crop-producing regions in the Mississippi Basin are wholly included in the USMP interpretation of the basin. Table 4.3 shows baseline conditions in the model for crop acreage, prices, fertilizer use, pollutant loss, and tillage practices.

TABLE 4.3. Baseline for the U.S. mathematical programming (USMP) model.

Crop Type	**Crop Acreage** (in millions) *Miss. Basin*	*Total U.S.*	**U.S. Crop Price**
Corn	64.7	81.3	$2.80/bu
Sorghum	9.0	10.9	$2.50/bu
Barley	3.8	7.2	$2.60/bu
Oats	4.0	4.7	$1.70/bu
Wheat	65.8	75.5	$4.30/bu
Rice	1.4	2.9	$10.31/cwt
Soybeans	50.3	62.3	$6.45/bu
Cotton	8.7	14.0	$331.20/bale
Silage	4.3	6.7	$21.69/ton
Hay	38.8	62.4	$60.49/ton
Total	**250.8**	**328.0**	

Fertilizer	**Fertilizer Use (+)/ Pollutant Loss (-)** *Miss. Basin*	*Total U.S.*
Nitrogen	+7.52/-5.18	+9.67/-6.61
Phosphorus	+3.51/-0.41	+4.58/-0.53
Sediment	-844.3	-1123.2

Tillage Method	**Millions of Acres** *Miss. Basin*	*Total U.S.*
Conventional	131.1	169.6
Moldboard	49.0	74.9
Mulch	43.8	50.0
No Till	25.5	32.2
Ridge Till	1.4	1.4

4.2 CROPLAND NITROGEN CONTROL

We used the model to perform two tasks. The first was to identify the level of nitrogen-loss reduction where agriculture starts to become significantly constrained, in terms of crop prices, acreage in production, and exports. Then we used the model to estimate the costs and effectiveness of different strategies for achieving an “acceptable” nitrogen-loss reduction. However, a policy cannot be based on nitrogen loss, since it cannot be observed. Instead, a policy must be based on those factors of production that can be observed, such as input use, rotations, and tillage practices. In either case, the model finds the optimal combination of rotations, conservation practices, tillage practices, and fertilizer application rates to meet the stated goal.

4.2.1 Parametric Reduction in Nitrogen

We used the USMP model to evaluate how the agricultural sectors both within the Mississippi Basin and nationally respond to constraints on nitrogen loss from cropland in the basin of 20%, 30%, 40%, 50%, and 60%. The USMP model calculates only nitrogen loss at the edge of the field or at the bottom of the root zone. We assumed that reducing edge-of-field or root-zone loss would generate a similar percentage reduction in nitrogen loadings to water resources from cropland. It is important to note that the nitrogen-loss reductions we modeled are generally beyond the range where farmers can reasonably be expected to be able to reduce fertilizer application rates without reducing physical yields.

The nitrogen-loss constraints we modeled are for the basin as a whole, and not for each acre. Using a basin-wide constraint reduces the cost of the policy over a per-acre constraint, in that it allows low-cost areas (in terms of reducing nitrogen loss by one unit) to contribute a greater share of achieving the environmental goal. We selected these constraints because they span a range of N-loss reductions from what is claimed to be feasible by proponents of “win-win” nutrient management policies (20%), to a constraint that would severely stress the sector (60%).

In response to the N-loss constraints, the USMP model adjusted crop rotations, tillage practices, and fertilizer inputs within the Mississippi Basin in order to meet the constraint while maximizing welfare. The model favors those crops and cropping practices that have low nitrogen “leakage.” Where the model cannot find a crop-production system that allows for positive net returns, the land is retired from production. Outside the basin, where there are no constraints on N loss, farmers are free to respond to price changes. Since the basin is such an important crop-producing region, changes in production are likely to have noticeable impacts on prices.

We first present the results in terms of the 20% scenario, and use these to highlight how the model performed. The 20% scenario produced relatively modest impacts. Within the basin, total crop acreage is reduced by about 6%. The impacts on crop acreage and prices at the national level are within the range seen over the period 1987–96 for all crops (Tables 4.4 and 4.5). Production outside the basin is stepped up in response to higher prices. The rest of the country cannot make up all the production losses in the basin, however, because of limitations on acreage and land that is generally not as productive. Consequently, prices rise for most crops, which reduces exports for most crops, but not significantly (Table 4.6).[3] Net social welfare (producer plus consumer surplus) is reduced by 0.08%, or about $830 million (Table 4.7).

[3]The USMP model results indicate that all the scenarios result in increased imports for some crops, but the impacts in percentage terms are much less than the impacts on exports.

TABLE 4.4. Percentage change in crop acreage as a result of reductions in N loss.

Area	Corn	Sor-ghum	Bar-ley	Oats	Wheat	Rice	Soy-beans	Cot-ton	Sil-age	Hay	Total
					20% Reduction in N Loss						
MB[1]	-1.3	-23.9	-4.6	-5.2	-8.5	-2.0	-4.2	-11.2	-2.1	-6.0	-5.8
RUS[1]	4.8	26.3	0	0	3.1	0	2.5	3.8	0	0.4	2.8
US	-0.1	-15.7	2.2	-4.4	-7.0	-0.9	-2.9	-6.0	-1.1	-3.5	-3.8
					30% Reduction in N Loss						
MB[1]	-4.5	-38.8	-12.8	-10.8	-20.6	-4.0	-10.9	-19.3	-5.2	-13.0	-13.3
RUS[1]	9.6	42.1	2.9	0	7.2	0	6.7	5.7	0	1.3	6.0
US	-1.6	-24.7	-5.6	-8.8	-17.1	-1.8	-7.5	-10.0	-2.5	-7.6	-8.8
					40% Reduction in N Loss						
MB[1]	-8.2	62.0	-19.3	-19.1	-31.0	-6.9	-19.1	-12.9	-9.0	-24.4	-21.3
RUS[1]	15.7	57.9	5.9	14.3	11.3	0	12.5	5.7	4.2	2.5	10.0
US	-3.3	-41.2	-7.3	-15.5	-25.6	-3.1	-13.0	-5.4	-4.2	-14.2	-14.0
					50% Reduction in N Loss						
MB[1]	-16.3	-64.6	-32.3	-26.5	-42.5	-14.9	-30.7	-10.2	-15.6	-46.9	-32.7
RUS[1]	23.5	68.4	11.8	14.3	14.4	0	18.3	9.4	8.3	3.8	14.4
US	-8.2	-40.8	11.0	-21.3	-35.1	-6.2	-21.1	-2.7	-7.7	-27.6	-21.6
					60% Reduction in N Loss						
MB[1]	-26.1	-78.2	-43.5	-39.1	-51.8	-40.6	-43.3	-10.7	-29.3	-63.3	-44.0
RUS[1]	32.5	89.4	17.6	14.3	18.6	6.7	27.5	11.3	12.5	5.5	19.4
US	14.1	-49.5	-14.9	-31.6	-42.8	-16.0	-29.8	-2.1	-15.2	-37.3	-29.1

[1]MB = Mississippi Basin; RUS = Rest of U.S.

TABLE 4.5. Percentage change in crop price as a result of reductions in N loss.

Crop	Percentage Reduction in N Loss				
	20	***30***	***40***	***50***	***60***
Corn	9.4	19.1	29.9	43.8	61.8
Sorghum	19.3	31.7	44.4	55.4	75.1
Barley	-0.5	2.6	9.0	18.5	24.3
Oats	-3.3	4.4	12.4	26.2	46.7
Wheat	5.4	14.1	24.4	32.6	40.2
Rice	1.3	2.4	3.4	5.9	13.7
Soybeans	2.5	6.6	11.7	21.0	34.2
Cotton	2.9	5.5	6.6	8.5	10.7
Silage	0.4	1.2	2.0	3.5	6.4
Hay	1.0	2.2	4.5	7.8	11.0

TABLE 4.6. Percentage change in export volume as a result of reductions in N loss.

Crop	Percentage reduction in N Loss 20	30	40	50	60
Corn	-5.3	-10.9	-17.0	-24.9	-35.3
Sorghum	-22.6	-37.1	-52.0	-64.8	-87.9
Barley	0.3	-1.7	-5.9	12.0	-15.8
Oats	2.1	-2.9	-8.0	-17.0	-30.4
Wheat	-8.8	-22.9	-39.6	-52.8	-65.2
Rice	-2.8	-5.0	-7.3	-12.5	-29.1
Soybeans	1.9	-5.2	-9.1	-16.3	-26.6
Cotton	3.8	-7.3	-8.7	-11.4	-14.2

TABLE 4.7. Summary of economic costs of N loss-reduction strategies.

Scenario	Social Welfare (millions of $)	N-Loss Reduction (1,000 metric tons)	Unit Cost ($/ton)
N-Loss Reduction			
20%	-831	941	883
30%	-2,677	1,412	1,,896
40%	-6,343	1,882	3,370
50%	-12,239	2,352	5,204
60%	-21,109	2,822	7,480
Fertilizer Reduction			
20%	-347	503	690
45%	-2,922	1,027	2,845
500% Fertilizer Tax	-14,932	1,027	14,539
Wetlands Acreage[1, 2]			
1,000,000	-406	67	6,060
5,000,000	-3,115	350	3,503
10,000,000	-7,537	713	10,571
18,000,000	-15,506	1,300	11,928
Buffer[3]	-18,014	692	26,032
20% Fert. Reduction in 5,000,000 Acres	-4,854	882	5,501

[1]Social welfare for wetland strategies includes changes in consumer and producer surpluses, plus wetland restoration costs.
[2]Wetland filtering capacity: 15 g/m^2.
[3]Buffer filtering capacity: 4 g/m^2.

Increases in net returns are not felt equally by crop and livestock sectors. The livestock sector purchases grain for feed, so the increase in prices hurts livestock producers. In looking at these two sectors, we had to deal with a minor shortcoming of the model. The income and cost data used to build the model cannot be disaggregated to the same level of geographic detail as the data on the physical aspects of agriculture. While the crop acreage, production, and environmental factors were all modeled on the basis of the basin, the resulting impacts on farm net returns can only be reported on the basis of the multi-state farm production regions. We selected the Corn Belt, Northern Plains, and Mississippi Delta regions as most representative of the basin, as defined in the USMP. These three regions contain over 70% of the cropland in the basin. Net farm income increases for crop producers in these regions when nitrogen losses are restricted (Figure 4.2). However, this does not mean that all farmers within the basin benefit. Those producers who have economically marginal cropland that is not profitable to farm given the N-loss constraints suffer a loss in net returns.

The impact of the 20% N-loss reduction constraint on environmental indicators shows the importance of considering all the impacts of a policy (Figures 4.3–4.5). Nitrogen losses within the basin are reduced 20%, as modeled. Adjustments within the basin also result in decreases in phosphorus loss and soil erosion. Each of these reductions also generates water quality benefits. However, the increases in crop prices result in more intensive production on existing acreage outside the basin, and in an overall increase in acreage in production. Without any environmental constraints, nitrogen loss, phosphorus loss, and soil erosion increase in the rest of the U.S., to the possible detriment of water quality.

As the constraint on nitrogen loss is further tightened, the economic impacts on the agricultural sector rapidly escalate. With a 30% reduction in N loss, crop prices increase above recent levels for most crops, and crop acreages are below recent levels. Over 20% of the crop acreage within the basin is retired. Exports are significantly reduced, particularly for wheat and corn. The negative environmental impacts outside the basin from increases at the intensive and extensive margins of agricultural production become more pronounced. For the 60% N-loss reduction constraint, the system is severely stressed, in terms of price increases and acreage changes. Prices are 15–50% higher than the highest prices seen over the past decade. Crop production ceases on 44% of the cropland in the basin. For a 60% reduction in N loss, social welfare is reduced by 2.15%, or $21 billion (Table 4.7).

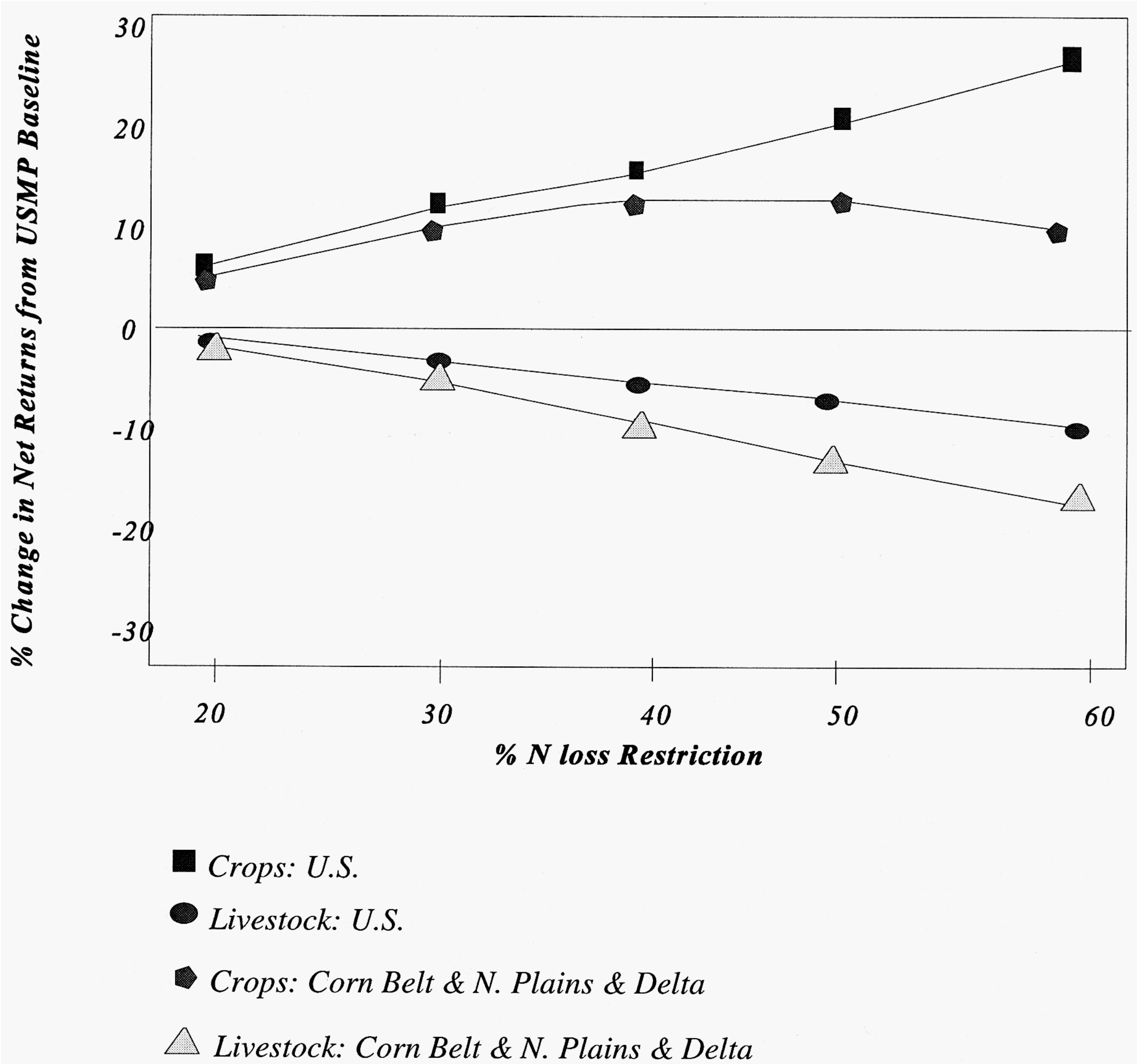

FIGURE 4.2. Impacts of nitrogen-loss restrictions on producer net returns.

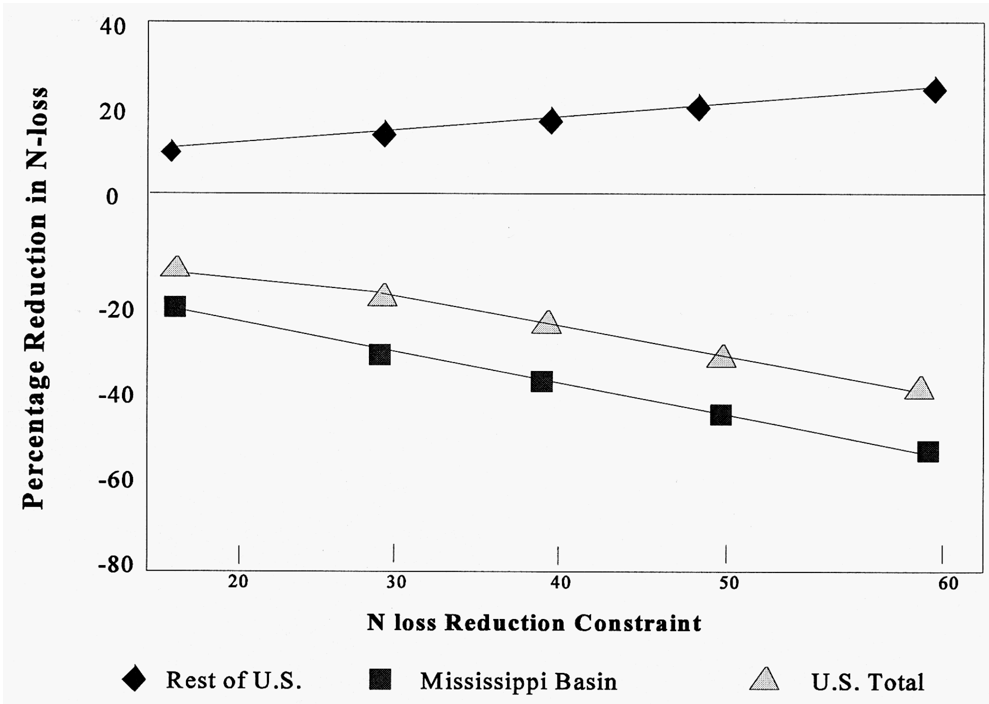

FIGURE 4.3. Impacts of nitrogen-loss restrictions on nitrogen loss.

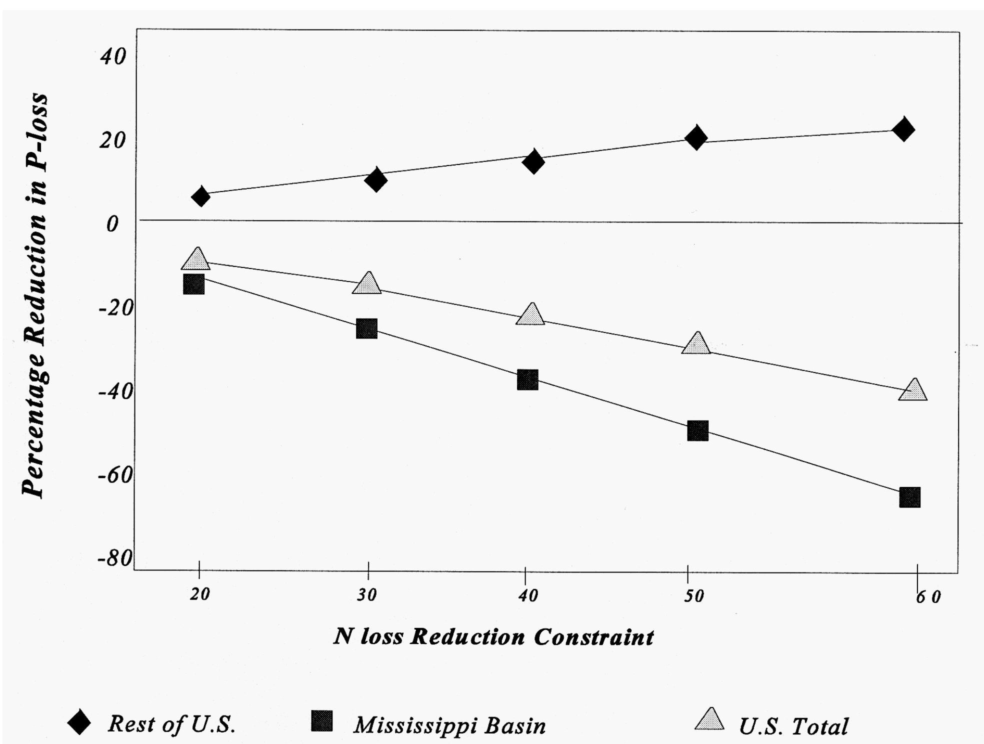

FIGURE 4.4. Impacts of nitrogen-loss restrictions on phosphorus loss.

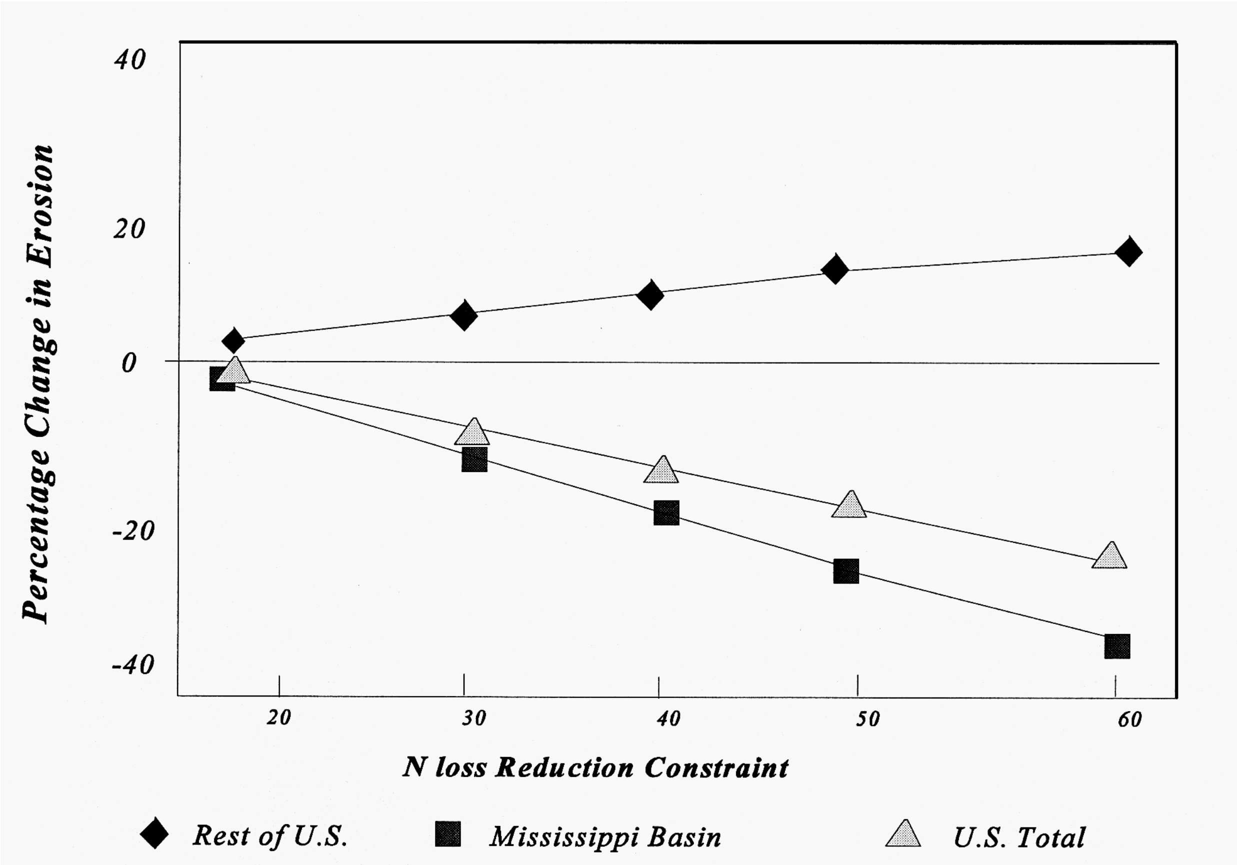

FIGURE 4.5. Impacts of nitrogen-loss restrictions on soil erosion.

4.2.1.1 ACHIEVING AN ENVIRONMENTAL GOAL

After reviewing these results, we selected the 20% reduction in edge-of-field N-loss scenario as the one that offers the best combination of sizable nitrogen-loss reductions and acceptable economic costs. The question now is: How can we achieve this goal with policy tools that are based on observable factors of production, such as fertilizer use, tillage practices, and crop rotations?

The results of the 20% N-loss constraint exercise give us the least-cost solution for achieving that goal, in terms of regional allocation of cropland, rotations, tillage practices, and fertilizer application rates. It is technically possible to achieve this solution, by "prescribing" optimal agronomic practices for every enterprise recognized by the model, either through regulation or through other types of incentive. However, the transaction costs (developing site-specific plans and monitoring all producers' activities) would almost certainly outweigh the benefits. In addition, the results of a model such as USMP are not detailed enough for a real application of the results to the ground. Instead, policy must be designed around a few factors that are easy to observe and that are closely related to nitrogen loss.

We reviewed the results of the N-loss restriction scenarios to try to identify those "drivers" that seemed to be most responsible for meeting the constraint. We determined that a combination of nitrogen fertilizer-use reductions and land retirement were most responsible. Fertilizer use appeared to be the most important factor in achieving a 20% N-loss reduction goal; tillage practices and rotations were not important factors in the USMP model for achieving this goal. Therefore, we focused on reducing nitrogen fertilizer use.

4.2.2 Restricting Fertilizer to Achieve the Nitrogen-Loss Goal

One way of achieving the N-loss goal is to restrict fertilizer use through a regulatory standard. We first ran the model with a basin-wide 20% restriction on fertilizer use. The reduction in fertilizer nitrogen use on crops in the 20% N-loss scenario was 24%, suggesting that a 20% reduction in N fertilizer may result in a sizable reduction in N loss. We let the model allocate the fertilizer-use reduction across the basin, rather than require each farmer to reduce fertilizer use by 20%. This would minimize the costs to the sector by allowing farmers who can most afford to reduce fertilizer use to provide a greater share of the reduction.

The results show what happens when nitrogen fertilizer use is the basis for a policy, rather than edge-of-field/bottom-of-root-zone nitrogen loss, which is the actual policy goal. Producers respond to the restriction by reducing fertilizer use and by shifting to crops and tillage practices that use less fertilizer, rather than taking those actions that necessarily reduce nitrogen loss. Under this scenario, production shifts toward nitrogen-fixing legume crops, such as soybeans, would be optimal behavior but would also tend to dampen the effectiveness of nitrogen-use restrictions.

The economic consequences of reducing fertilizer use by 20% are less pronounced than the 20% N-loss restriction. Farmers can apparently meet the constraint fairly painlessly by reducing purchased fertilizer use and replacing it with nitrogen fixation from legumes. As shown in Table 4.8, crop acreage is reduced by only 2% in the basin and by only 1% for the nation as a whole. Increases in commodity prices are about half those seen in the 20% N-loss constraint scenario. In Table 4.9, net returns to the crop sector increase in the basin's three major farm production regions, and exports decrease for all crops but soybeans. Net social cost declines by 0.04%, or $347 million (Table 4.7).

TABLE 4.8. Comparison of fertilizer-reduction strategies in the Mississippi Basin (MB).

Crop	Percent Change in Acreage in MB			Percent Change in Price		
	20% Fertilizer Constraint	*45% Fertilizer[1] Constraint*	*Combined*	*20% Fertilizer Constraint*	*45% Fertilizer[1] Constraint*	*Combined*
Corn	0.9	-5.9	1.8	5.5	28.2	9.2
Sorghum	-6.6	-29.9	-6.9	6.4	35.5	10.6
Barley	-2.4	-15.7	-3.4	1.8	12.2	3.3
Oats	-1.3	-7.8	-3.0	2.9	14.3	7.5
Wheat	-3.3	-14.5	-4.9	2.3	13.4	3.5
Rice	-1.7	-8.2	-7.1	1.0	3.3	2.8
Soybeans	0.0	0.2	-1.7	0.2	-1.3	2.0
Cotton	-13.2	-29.3	-15.3	2.3	7.0	4.0
Silage	-2.1	-11.6	-4.6	0.9	4.0	1.5
Hay	-0.5	-2.4	-3.5	0.1	0.3	0.7
Total	**-2.0**	**-8.4**	**3.6**			

[1]Price and acreage for changes for 500% nitrate tax the same.

TABLE 4.9. Comparison of economic impacts of fertilizer constraints and taxes.

Farm Production Region and Crop Type	20% Fertilizer Constraint	45% Fertilizer Constraint	500% Tax	Combined
	Percentage Change in Net Returns—Crops			
Corn B., North. Pl., Delta[1]	4.2	14.3	-14.3	6.7
Total U.S.	4.6	16.8	-8.4	7.4
	Percentage Change in Net Returns—Livestock			
Corn B., North. Pl., Delta[1]	-1.6	-8.0	-8.0	-2.7
Total U.S.	-0.9	-4.6	-4.6	-1.4
	Percentage Change in Export Volumes			
Corn	-3.1	-16.1	-16.1	-5.3
Sorghum	-7.5	-41.5	-41.5	-12.4
Barley	-2.7	-17.1	-17.1	-4.6
Oats	-1.9	-9.3	-9.3	-4.9
Wheat	-4.3	-25.2	-25.2	-6.6
Rice	-2.2	-6.6	-6.6	-6.1
Soybeans	0.1	1.0	1.0	-1.5
Cotton	-3.1	-9.2	-9.2	-5.3

[1]*Corn Belt, Northern Plains, and Delta farm production regions*.

Fertilizer use is serving as a proxy for, but it is not perfectly correlated with, nitrogen loss. N-loss reductions are less than might have been expected. Farmers shift to rotations that include soybeans, which is a legume and produces its own nitrogen. Soybean acreage increases relative to other crops. Nitrogen can leak from these rotations, even though purchased fertilizer use is reduced. Reducing nitrogen fertilizer use by 20% only reduced nitrogen loss by 10.7% in the basin—about half the desired goal (Table 4.10). Soil erosion increases in both the basin and the U.S., indicating that more erosive practices are being employed on existing cropland, or that more cropland is being placed in production.

TABLE 4.10. Impacts of fertilizer reduction on environmental indicators.

Indicators	20% Fertilizer Constraint			45% Fertilizer[1] Constraint			Combined		
	MB[2]	***RUS[2]***	***US***	***MB[2]***	***RUS[2]***	***US***	***MB[2]***	***RUS[2]***	***US***
N Loss	-10.7	1.7	-8.0	-21.8	7.6	-15.5	-18.8	2.2	-14.2
P Loss[3]	-3.9	1.6	-2.7	-9.4	5.8	-6.0	-5.5	1.7	-3.9
Erosion[3]	1.6	1.0	1.5	2.6	4.6	3.1	1.7	1.3	1.6

[1]*500% N fertilizer tax has identical results.*
[2]*MB = Mississippi Basin; RUS = Rest of U.S.*
[3]*For P loss and sediment, we did not estimate wetland filtering. Reductions are strictly edge-of-field from the Erosion Productivity Impact Calculator (EPIC) model.*

What level of fertilizer reduction will generate a 20% reduction in N loss? Results from the model indicate that a 45% reduction in fertilizer use will decrease N loss by 20%. Tables 4.8–4.10 summarize the impacts of a mandated 45% basin-wide reduction in nitrogen fertilizer use. The impacts on acreage and prices are much more pronounced than the 20% reduction in N-loss constraint, except for soybeans. Soybean acreage actually increases as soybeans are included in rotations to provide nitrogen to other crops. Export volumes are reduced for all crops, except soybeans. Net returns increase for the crop sector. Total social welfare declines by 0.3%, or $2.9 billion, which is more than for the cost-effective 20% N-loss scenario (Table 4.7). The difference in social costs can be viewed as the cost of not being able to base policy directly on nitrogen loss.

The environmental impacts of the 45% fertilizer constraint are more pronounced than for the 20% fertilizer constraint (Table 4.10). Nitrogen loss within the basin is reduced, but increases by 7.6% in the rest of the U.S. Soil erosion increases both within and outside basin. Erosion increases in the basin despite the reduction in cropland in production, indicating that more erosive practices and rotations are being used than under the cost-effective solution. In contrast, soil erosion in the basin decreased in the least-cost 20% N-loss scenario.

4.2.3 Taxing Fertilizer to Achieve the Nitrogen-Loss Goal

An alternative to a regulation to reduce N fertilizer use is to place a tax on fertilizer. A tax raises the price of fertilizer relative to other inputs, spurring farmers to find ways of reducing its use. Some states already use fertilizer taxes to raise revenue, but the low tax levels coupled with an inelastic demand for fertilizer ensure only negligible impacts on use.

The results from the USMP model indicate that a tax of 500% on nitrogen fertilizer would be required to achieve a 45% reduction in fertilizer use (and an approximate 20% reduction in N loss). The magnitude of the tax is supported by the finding that the demand for nitrogen fertilizer is inelastic (Vrooman and Larson 1991; Fernandez–Cornejo 1993). The impacts on crop production, crop prices, input use, and environmental indicators are identical to the 45% reduction standard (Tables 4.8–4.10). The only difference is in net returns to crop farmers and social welfare (Table 4.7). Because of the transfer of tax revenue to the government, net returns to producers are lower than for a fertilizer standard. Social welfare is reduced by 1.5%, or $14.9 billion (Table 4.7). Net returns in the Corn Belt, Northern Plains, and Delta farm production regions are reduced by 14.3% (Table 4.9). If the revenue from the tax is returned to producers as a reward to adopting nutrient management practices, the welfare impacts are the same for taxes as for the standard.

4.3 WETLAND RESTORATION FOR DENITRIFICATION

An alternative to reducing N loss at the source in agricultural fields is to intercept N in streams using restored wetlands that will cause denitrification. Table 4.2 of the Topic 5 report discusses restoring wetlands as an option to trap and denitrify surface- and ground-water flows into the Gulf. The report calculated that 52,267–73,173 km^2 (13–18 million acres) of additional wetlands in the Mississippi Basin would be required to reduce total N discharge to the Gulf by 50–70%. We used the screening process described above, along with the USMP model, to identify the constraints on agriculture—in terms of crop prices, acreage in production, and exports—caused by successively higher levels of wetland restoration.

In this section, we estimate the direct costs and agricultural economic impacts of alternative programs to restore 4,047 km^2, 20,234 km^2, 40,468 km^2, and 72,844 km^2 (1, 5, 10, and 18 million acres) of cropland formerly converted from wetlands. Total direct costs for permanent easements and restoration of wetlands selected at least cost range from $495 million to $32.4 billion (Table 4.11). Restoration costs range from 75% to 24% as enrollment increases. Easement costs are $312–$3,377 per hectare of wetlands, while restoration costs remain about $1,000 per hectare.

TABLE 4.11. Direct costs of wetland restoration, enrollment at least cost.

Direct Costs	**Wetland Area Restored**			
	4,047 km² ***(1 mill. acres)***	***20,234 km²*** ***(5 mill. acres)***	***40,469 km²*** ***(10 mill. acres)***	***72,844 km²*** ***(18 mill. acres)***
Program Costs *(millions of $)*	**495**	**4,694**	**13,732**	**32,361**
Easement Cost	126	2,531	9,023	24,601
Restoration Cost	369	2,163	4,709	7,760
Unit Costs—$/hectare ($/acre)	**1,224 (495)**	**2,320 (939)**	**3,393 (1,373)**	**4,443 (1,798)**
Easement Cost (including upland)	312 (126)	1,251 (506)	2,230 (902)	3,377(1,367)
Restoration Cost/Wetland Area	912 (369)	1,069 (433)	1,164 (471)	1,065 (431)

We used the USMP model to evaluate how the agricultural sector, both within the Mississippi Basin and nationally, responds to retiring cropland for wetland restorations. Cropland retired from production to restore wetlands is subtracted from the cropland used in the USMP model's baseline solution in each farm producing region. The amount of land retired is adjusted for differences in productivity between the average cropland acreage in the producing region and the productivity of the cropland identified as most likely to be restored to wetland in the screening step described above. In general, cropland likely to be restored to wetland is less productive than average cropland. Reducing cropland reduces production, which leads to increased prices. Higher prices, in turn, cause additional land to enter production, both within the basin and in the rest of the U.S. Acreage, price, and net return results presented below reflect the new equilibrium reached as a result of the original acreage reduction.

To match a 20% reduction in N losses from the USMP baseline level, 18 million acres of wetlands would have to be restored. Nitrogen losses in the USMP baseline are to the edge of the field and the bottom of the root zone, and are not directly comparable to the 1.5 million metric tons of N losses to the mouth of the basin reported in Table 4.2 of Topic 5. Therefore, the 20% loss is roughly equivalent to a 70% reduction in N losses to the mouth of the basin. Differences in the magnitude of impacts with fewer wetlands restored are also depicted and discussed.

A 72,844-km^2 (18-million-acre) increase in wetland restoration amounts to nearly 20 times more acreage than the existing Wetland Reserve Program, focused exclusively on the Mississippi Basin. This scenario produced relatively large impacts on the agricultural economy. The impacts on crop acreage and prices at the national level are much greater than the range seen over the period 1987–96, for all crops (Tables 4.12 and 4.13). Crop acreages generally decline by 3–20% in the basin, and U.S. crop prices increase by 2–18%. There is a 1% increase in production outside the basin overall in response to higher prices, but major crop acreage increases by 1–4%, while silage and hay acreage decreases. Higher prices reduce exports for most crops, ranging from 2% to 12% (Table 4.14).

TABLE 4.12. Percentage change in crop acreage, enrollment at least cost.

Crop	Wetland Area Restored											
	4,047 km^2 (1,000,000 acres)			*20,234 km^2 (5,000,000 acres)*			*40,469 km^2 (10,000,000 acres)*			*72,844 km^2 (18,000,000 acres)*		
	MB[1]	*RUS*[1]	*US*	*MB*[1]	*RUS*[1]	*US*	*MB*[1]	*RUS*[1]	*US*	*MB*[1]	*RUS*[1]	*US*
Corn	-0.1	0.0	0.0	-0.6	0.6	-0.3	-1.6	1.8	-1.0	-4.7	3.0	-2.7
Sorghum	-0.3	0.0	-0.2	-1.1	0.0	-0.7	-0.4	0.0	0.2	2.5	10.5	2.4
Barley	-0.4	0.0	-0.2	-1.6	0.0	-0.7	-2.0	0.0	-0.7	-1.9	2.9	-0.5
Oats	-0.2	0.0	-0.2	-1.5	0.0	-1.2	-3.0	0.0	-2.5	-5.8	0.0	-4.9
Wheat	-0.4	0.0	-0.3	-1.6	1.0	-1.4	-2.3	0.0	-2.0	-3.8	1.0	-3.1
Rice	-0.1	0.0	0.0	-4.8	0.0	-2.1	-10.4	0.0	-4.5	-19.3	0.0	-6.7
Soybeans	-0.1	0.0	0.0	-1.0	0.8	-0.7	-2.8	0.8	-2.1	-7.7	2.5	-5.0
Cotton	-0.2	0.0	-0.1	-1.1	0.0	-0.5	-1.8	1.9	-0.7	-3.2	3.8	-1.1
Silage	-0.2	0.0	-0.2	-1.6	0.0	-1.7	-3.5	0.0	-3.6	-8.4	-8.3	-8.1
Hay	-0.3	0.0	-0.2	-2.0	0.0	-1.3	-4.6	-0.4	-3.0	-12.0	-0.8	-6.5
Total	**-0.2**	**0.1**	**-0.2**	**-1.3**	**0.1**	**-0.9**	**-2.6**	**0.5**	**-1.9**	**-6.0**	**1.0**	**-3.8**

[1]MB = Mississippi Basin; RUS = Rest of U.S.
Source: U.S. Mathematical Programming Model analysis, Economic Research Service, USDA.

TABLE 4.13. Percentage change in crop prices, enrollment at least cost.[1]

Crop	Wetland Area Restored			
	4,047 km^2 (1,000,000 acres)	*20,234 km^2 (5,000,000 acres)*	*40,469 km^2 (10,000,000 acres)*	*72,844 km^2 (18,000,000 acres)*
Corn	0.22	1.77	4.90	11.23
Sorghum	0.31	2.08	4.71	9.98
Barley	0.22	1.04	1.97	3.92
Oats	0.30	2.10	6.66	17.88
Wheat	0.26	1.14	1.82	3.08
Rice	0.03	1.60	3.43	5.14
Soybeans	0.07	1.21	3.99	8.72
Cotton	0.06	1.22	2.54	4.06
Silage	0.06	0.45	0.95	2.22
Hay	0.06	0.37	0.88	1.94

Source: U.S. Mathematical Programming Model analysis, Economic Research Service, USDA.

TABLE 4.14. Percentage change in crop exports, enrollment at least cost.[1]

Crop	Wetland Area Restored 4,047 km^2 (1,000,000 acres)	20,234 km^2 (5,000,000 acres)	40,469 km^2 (10,000,000 acres)	72,844 km^2 (18,000,000 acres)
Corn	-0.1	-1.0	-2.8	-6.4
Sorghum	-0.4	-2.4	-5.5	-11.7
Barley	-0.3	-1.4	-2.9	-2.5
Oats	-0.2	-1.4	-4.3	-11.6
Wheat	-0.5	-2.1	-3.4	-5.0
Rice	-0.1	-3.4	-7.3	-10.9
Soybeans	-0.1	-0.9	-3.1	-6.8
Cotton	-0.1	-1.6	-3.4	-5.4

[1]*Includes commercial exports and exports under export enhancement programs.*
Source: U.S. Mathematical Programming Model analysis, Economic Research Service, USDA.

Net farm income increases overall in the Corn Belt, Northern Plains, and Delta (representing the entire basin), as well as for the U.S. (Table 4.15). Increases in crop prices more than make up for the reduction in output. Increases in net returns are not felt equally by crop and livestock sectors. The livestock sector purchases grain for feed, so the increase in prices hurts livestock producers inside the basin as well as in the U.S. The increase in crop prices also hurts consumers by reducing consumer surplus. In total, net national social welfare (producer plus consumer surplus, plus wetland restoration costs) is reduced by 1.5%, or $15.5 billion (Table 4.7).

TABLE 4.15. Percentage change in farm net cash returns and net social benefit, enrollment at least cost.[1]

Type of Enterprise	Wetland Area Restored 4,047 km^2 (1 million acres)	20,234 km^2 (5 million acres)	40,469 km^2 (10 million acres)	72,844 km^2 (18 million acres)
Crop Enterprises				
CB, NP, Delta[1]	0.2	1.2	3.1	7.2
Total U.S.	0.2	1.4	3.7	7.8
Livestock Enterprises				
CB, NP, Delta[1]	-0.1	-0.6	-1.8	-3.9
Total U.S.	0.0	-0.3	-1.0	-2.1
Total Farm Net Cash Return–U.S.	0.1	0.9	2.4	5.5
Total Net Social Benefit[2]–U.S.	-0.04	-0.31	-0.74	-1.50

[1]Corn Belt, Northern Plains, Delta farm production regions.
[2]Change in farm net cash income and consumer prices for food and fiber products.
Source: U.S. Mathematical Programming Model analysis, Economic Research Service, USDA..

The impact of the 72,884-km^2 wetland restoration on environmental indicators shows the importance of considering all the impacts of a policy (Table 4.16). Nitrogen losses avoided on the restored wetlands, net of increases from new acres brought into production, within the basin are reduced by more than 173,000 metric tons. Additionally, and more important, is the estimated 728,424–1,092,636 metric tons in annual denitrification of surface and ground waters flowing through the restored wetlands. The total reduction from both sources is 19.9–27.6% of USMP baseline N losses. These two classes of N-loss reductions technically cannot be directly compared. Reductions in nitrogen associated with sediment, leached, and dissolved in runoff are only estimated at the edge of the field and bottom of the root zone and are not adjusted for fate and transport to the stream system. Estimates of denitrification in wetlands are from the nitrogen delivered to streams. However, the sum of the N-loss reductions can be considered as a relative measure of total reductions achieved by each wetland restoration level because the fate and transport adjustments needed for each restoration level are likely of the same order of magnitude.

Adjustments within the basin also result in decreases in phosphorus losses and soil erosion. Each of these reductions also generates water quality benefits. However, the increases in crop prices result both in more intensive production on existing acreage outside the basin, and in an increase in acreage in production. Without any environmental constraints, soil erosion increases in the rest of the U.S., to the probable detriment of water quality.

As less cropland acreage is restored to wetlands, economic impacts on the agricultural sector decrease. With restoration of 4,047–40,468 km^2 (1–10 million acres) of wetlands, total crop acreage in the basin decreases from 0.2% to 2.6% (Table 4.12). Offsetting increases in the rest of the U.S. are insufficient to make up the losses, resulting in a 0.2–1.9% decrease in U.S. crop acreage.

TABLE 4.16. Change in nutrient and sediment losses, enrollment at least cost.

Area/Type of Loss	Wetland Area Restored			
	4,047 km^2 (1 million acres)	***20,234 km^2 (5 million acres)***	***40,469 km^2 (10 million acres)***	***72,844 km^2 (18 million acres)***
	Metric Tons Lost to Field Boundary[1] in the Mississippi Basin			
N Loss	-5,443	-40,823	-90,718	-173,272
P Loss	-634	-4,536	-9,979	-17,237
Erosion	-1,470,547	-7,483,367	-15,653,473	-29,478,968
	Metric Tons of N Trapping[2] in the Mississippi Basin			
Low	-40,470	-202,340	-404,680	-728,424
High	-60,705	-303,510	-607,020	-1,092,636
	Metric Tons Lost to Field Boundary[1] in the Rest of the U.S.			
N Loss	0	3,629	9,072	19,051
P Loss	0	0	907	908
Erosion	50,802	454,500	1,275,502	2,360,495
	Metric Tons Lost to Field Boundary[1] in the Entire U.S.			
N Loss	-5,443	-37,195	-81,647	-154,221
P Loss	-634	-4,536	-9,072	-16,329
Erosion	-1,419,744	-7,028,867	-14,377,971	-27,118,473

[1]*Estimated losses to edge of field or bottom of the root zone based on EPIC simulations.*
[2]*Based on an estimated 10–15g-N/m2/yr (Table 4.2 of Mitsch et al. 1999).*
Source: U.S. Mathematical Programming Model analysis, Economic Research Service, USDA.

Rice, hay, silage, and soybeans are the major crops affected. Resulting crop price increases range from negligible to 7%, with the largest increases in oats, corn, and soybeans (Table 4.13). Export volumes decline by as much as 7% for sorghum, oats, and rice (Table 4.14). Farm net cash income for the U.S. increases by as much as 2.4%, with up to 4% increases for crops and decreases for livestock net income (Table 4.15). Higher consumer prices offset farm net cash income gains, resulting in net social benefits decreasing by up to 0.74%. Nitrogen, phosphorus, and sediment losses from reduced crop production and wetland trapping in the basin are proportionally reduced as fewer wetlands are restored, ranging from 1% to 15% of baseline levels. However, offsetting losses in the rest of the U.S. also increase (Table 4.16). Both 4,047 km^2 and 20,234 km^2 (1–5 million acre) restoration levels are probably feasible without unduly constraining the agricultural economy, but levels beyond this have significant impacts.

4.3.1 Targeting Wetland Enrollment

An alternative wetland targeting strategy is to target restoration on the basis of regional nitrogen loads, rather than on the cost of land retirement. An extra reduction in nitrogen filtering could make such a targeting strategy more favorable on a per-unit N-reduction cost basis. Based on the previous analysis of the agricultural impacts from different levels of wetland restoration, we chose a 20,234-km^2 (5-million-acre) level for further analysis. In addition, we examined the mechanism for achieving such a level of restoration, direct and indirect costs, and nitrogen reductions.

A program similar to the current Wetland Reserve Program (WRP), operated by USDA's Natural Resources Conservation Service (NRCS), is assumed to be the mechanism by which these lands are restored as wetlands. Landowners who choose to participate in the WRP sell a conservation easement to USDA to restore and protect wetlands. The landowner voluntarily limits future use of the land, yet retains private ownership. The landowner and NRCS develop a plan for restoring and maintaining the wetland. Permanent easements are conservation easements in perpetuity. In the current program, easement payments are the lesser of: the agricultural value of the land, an established payment cap, or an amount offered by the landowner. In addition to paying for the easement, USDA pays 100% of the costs of restoring the wetland. The current WRP has an enrollment cap of 3,946 km^2 (975,000 acres), and about 2,100 km^2 (533,000 acres) have been enrolled (USDA 1997).

In the absence of explicit criteria for enrolling restorable cropland to maximize nitrogen reduction, we assumed two scenarios: (1) the acreage enrolled is proportional to the total nitrogen yield by hydrologic unit, and (2) the acreage is enrolled at least cost from hydrologic units that yield some nitrogen. The geographic pattern of enrolling 5 million acres under the two scenarios is shown in Figure 4.6. Enrolling proportional to nitrogen yield produces a more uniform distribution than enrolling at least cost, which is concentrated in watersheds with cropland on wetland soils, with poor drainage, or low productivity. The more concentrated enrollment in the least-cost pattern raises the possibility of diminishing marginal nitrogen reduction, as more and more land is enrolled in the same watersheds. On the other hand, by widely distributing enrollment, the proportional enrollment pattern may be more likely to maximize the amount of drainage water filtered by the restored wetlands.

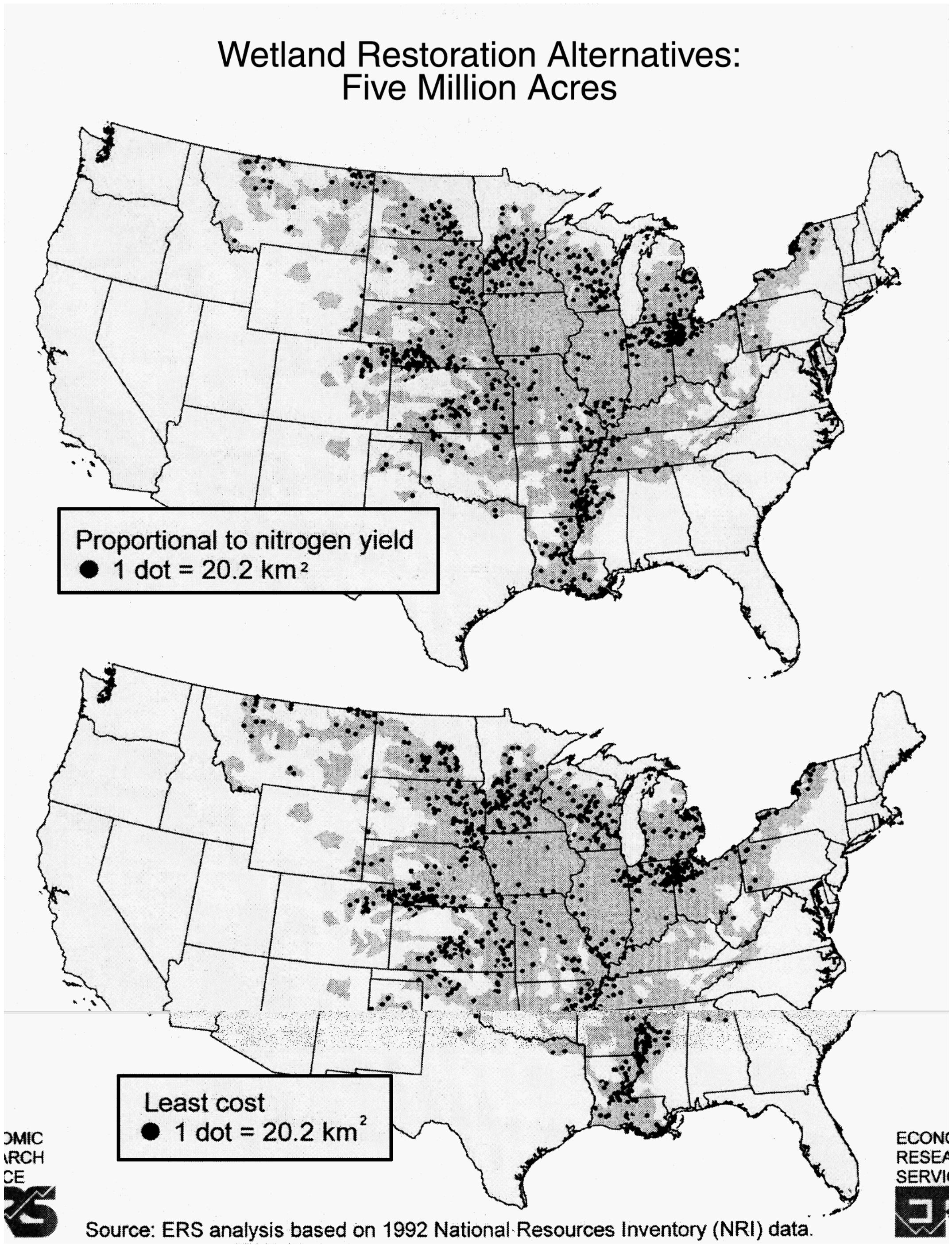

FIGURE 4.6. Wetland restoration alternatives. (NOTE: Some wetland sites in the lower Great Lakes were inadvertently brought into the solution. For the 1 million acres of wetland, this probably slightly lowered the cost of this solution.)

Total direct costs are significantly higher than least cost when restored land is drawn in proportion to nitrogen yields. Nitrogen yields tend to be high in areas where cropland is more productive, leading to higher opportunity costs. Easement and restoration costs under the proportional scenario rise from $1.3 billion to $39.5 billion as wetland acreage enrolled increases. Costs per wetland hectare restored increase from $3,153 to $5,419. Easement costs per hectare double as land enrolled under the proportional scenario increases, while restoration costs remain nearly constant at about $1,000 per hectare.

A comparison of the performance of a 20,234-km^2 (5-million-acre) wetland restoration level enrolled proportional to nitrogen yields and at least cost is detailed in Table 4.17. Indirect costs, in terms of changes in farm net cash income and consumer surplus due to changes in crop prices, are higher for proportional enrollment. Field N reductions are about 25% higher. It is not known how much different nitrogen trapping and denitrification in wetlands enrolled proportional to N loadings would be from least-cost enrollment. Assuming equal levels (at 10 g per m^2), least-cost enrollment reduces USMP baseline N loss by 5.3%, compared with 5.7% for proportional enrollment. Direct costs per kg of N reduction are $3.02 for proportional enrollment and $1.92 for least-cost enrollment. If the sum of direct and indirect costs is considered, proportional enrollment costs $17.50 per kg reduced per year, compared with $14.32 per kg for least-cost enrollment. Based on this analysis, further discussions of wetland restoration will be based on the least-cost targeting strategy.

TABLE 4.17. Performance of a 20,234-km^2 (5-million-acre) wetland restoration program.

Performance Indicators	Enrolled at Least Cost		Enrolled Proportional to N Yield	
	Total (millions of $)	**$/Hectare ($/Acre)**	**Total (millions of $)**	**$/Hectare ($/Acre)**
Annual Direct Costs	**114**	**29 (23)**	**126**	**30 (25)**
Restoration	101	8 (20)	100	8 (20)
Administrative[1]	14	1(3)	27	2 (5)
Annual Indirect Costs	**243,163**	**19.6**	**303,510**	**24.4**
Farm Cash Income[2]	-586	-47 (-117)	-921	-74 (-183)
Consumer Surplus	3,600	290 (716)	5,314	428 (1,057)
Net Social Benefit	3,014	242 (599)	4,394	353 (873)
	Metric Tons	**Kg**	**Metric Tons**	**Kg**
N Reductions[3]	**243,163**	**19.6**	**303,510**	**24.4**
Field Losses[4]	40,823	3.3	58,967	4.7
Trapping[5]	202,340	16.3	202,340	16.3
P Reductions	**4,536**	**0.4**	**6,350**	**0.5**
Sediment Reduction	**7,483,367**	**602.1**	**9,355,796**	**752.7**

[1]Three percent of easement and restoration costs (Misso, personal communication).
[2]Includes the opportunity cost of land idled to restore wetlands, compensated by easement cost paid to landowners.
[3]Field losses and trapping are not strictly additive.
[4]Estimated losses to the edge of the field or the bottom of the root zone are based on EPIC simulations.
[5]Based on an estimated 10–15 g-N/m^2/yr (Table 4.2 of Mitsch et al. 1999).
Source: U.S. Mathematical Programming Model analysis, Economic Research Service, USDA.

4.4 RIPARIAN BUFFERS FOR NITROGEN FILTERING AND DENITRIFICATION

Another strategy to intercept N runoff from agricultural fields and other nonpoint sources before it reaches streams is to convert land bordering stream edges into vegetated riparian buffers that trap nutrients and sediment, and to promote denitrification in the root zone of the buffer vegetation. Section 3.2.4 of Topic 5 discusses the construction and efficiency of such buffers. Because the nitrogen removal efficiency of riparian buffers is about one-fourth that of wetlands, the Topic 5 team calculated that 196,000–274,400 km^2 (48–68 million acres) of riparian buffers in the Mississippi Basin would be needed to reduce basin nitrogen loads by 50–70% (Table 4.3).

At an average width of 30 meters (100 feet) on each side of a stream, or 24 acres per linear mile, 2–3 million miles of riparian buffers would translate into some 24–36 million acres, larger than the entire 7-million-acre goal of the USDA Stream Buffer Initiative (NRCS web site).

Based on the screening process for restorable wetland sites described above, along with the USMP model, we identified the constraints on agriculture—in terms of crop prices, acreage in production, and exports—caused by a comparable level of riparian buffer restoration. In this section, we estimate the direct costs and agricultural economic impacts of a program to restore 109,266 km^2 (27 million acres) of riparian buffers, involving 18 million acres of cropland, enrolled in areas proportional to nitrogen yields. This level of riparian buffer restoration is expected to reduce USMP baseline N losses by 15% through a combination of field reductions from land taken out of crop production and denitrification of intercepted runoff and base flow by the buffer strip vegetation. This corresponds to about 44% of N losses to the mouth of the basin, as estimated in Table 4.3 of Topic 5. As the Topic 5 authors stress, the location and nature of the riparian buffers needed would have to be chosen with a precision that neither team could model. Specifically, if the location within the producing regions of the USMP model of the riparian buffer areas needed differs substantially from that of the wetland restoration areas identified in the screening process, the overall economic impacts could differ from what is estimated here.

Total direct costs for permanent easements and restoration of riparian buffers are estimated at $46.3 billion (Table 4.18). Easement costs are estimated at $4,348 per hectare of cropland. However, because cropping does not generally extend directly to the water's edge, we assumed that one-third of the buffer area would not be cropped and would have an opportunity cost of zero, implying zero easement cost. Restoration costs assume that 40% of the buffer areas, mostly in the Northern Plains, will be planted to grass at the average cost incurred to establish such buffers in the Conservation Reserve Program in 1996, while the remaining 60%, mostly in the Corn Belt and Delta regions, will be planted to trees. The total buffer area of the will be planted, despite the assumption that only two-thirds of it is cropland that will be paid compensation.

We used the USMP model to evaluate how the agricultural sector, both within the Mississippi Basin and nationally, responds to retiring cropland for riparian buffer restorations. Cropland retired from production to restore buffer strips is subtracted from the cropland used in the USMP model's baseline solution in each producing region. The amount of land retired is adjusted for differences in productivity between the average cropland acreage in the producing region and the productivity of the cropland identified as most likely to be restored to buffers, as in the screening step described above. In general, cropland likely to be restored to buffers is less productive than average cropland because of drainage problems. Reducing cropland reduces production, which leads to higher prices. Higher prices, in turn, cause additional land to enter production, both within the basin and in the rest of the U.S. Acreage, price, and net return results presented below reflect the new equilibrium reached as a result of the original acreage reduction.

TABLE 4.18. Direct costs of wetland restoration, enrollment at least cost.

Area/Types of Costs	Cropland	Stream Margin	Total
Area[1]—*km² (millions of acres)*	**72,844 (18)**	**36,422 (9)**	**109,266 (27)**
Program Costs —*millions of $*	**38,998**	**3,664**	**42,662**
Easement Cost	31,670	0	31,670
Restoration Cost[2]	7,328	3,664	10,993
Unit Costs—$/hectare ($/acre)	**5,354 (2,167)**	**1,006 (407)**	**4,240 (1,389)**
Easement Cost	4,348 (1,759)	0	3,234 (982)
Restoration Cost	1,006 (407)	1,006 (407)	1,006 (407)

[1]Assumes that one-third of the buffer is stream margin that is not cropped and thus has zero opportunity cost.
[2]Assumes that 40% of buffers will be planted to grass at an average cost of $170 per acre, based on 1996 CRP filter strip costs for selected states, and 60% will be planted to trees at an average cost of $565 per acre.

A 109,622-km^2 (27-million-acre) increase in riparian buffer restoration amounts to buffering 1.1 million miles of streams, more than the goal of the USDA Conservation Buffer Initiative, focused entirely on the Mississippi Basin. This scenario produced rather severe impacts on the agricultural economy. The impacts on national crop acreage and prices are greater than those seen during 1987–96 for all crops (Table 4.19). Crop acreages generally decline by 2–20% in the basin and 1–8% in the U.S. Crop prices increase by 2–21%. Production increases substantially outside the basin in response to higher prices, which in turn reduce exports significantly for most crops.

Net returns to crop producers in the Corn Belt, Northern Plains, and Delta (representing the basin) increase by 5.8% for these regions, as well as 4.9% for the U.S. (Table 4.19). Increases in crop prices more than make up for the reduction in output, except in the Delta region. Increases in net returns are not felt equally by the crop and livestock sectors. Higher prices for feed grain hurt livestock producers both inside and outside the basin. Higher crop prices also hurt consumers by reducing the consumer surplus. In total, net national social welfare (producer plus consumer surplus) is reduced by 1.8%, or $18 billion.

The impact of the 109,266-km^2 riparian buffer restoration on environmental indicators shows the importance of considering all the impacts of a policy (Tables 4.19 and 4.20). Nitrogen losses avoided on the restored riparian buffers, net of increases from new acres brought into production, within the basin are reduced by almost 255,000 metric tons, as modeled. Additionally, and more important, is the estimated 437,000 metric tons in annual denitrification of surface and ground waters flowing over and through the restored riparian buffers. This is 15% of USMP baseline N losses to the edge of the field and the bottom of the root zone, and 44% of N losses to the mouth of the basin, as estimated in Table 4.3 of the Topic 5 report.

These two classes of nitrogen-loss reductions technically cannot be directly compared. Reductions in nitrogen associated with sediment, leached, and dissolved in runoff are only estimated at the edge of the field and bottom of the root zone and are not adjusted for fate and transport to the stream system. Estimates of filtering and denitrification in riparian buffer areas are from the nitrogen intercepted on the way to streams. However, the sum of the nitrogen-loss reductions can be considered as a relative measure of total reductions achieved by different approaches and restoration levels because the fate and transport adjustments needed for each are likely of the same order of magnitude.

TABLE 4.19. Change in crop acreage, prices, returns, and nutrient losses for a 109,266-km^2 (27-million-acre) riparian buffer restoration program enrolled proportional to nitrogen yield.

Indicators	Miss. Basin	Rest of U.S.	Total U.S.	Total U.S.
	Percent Change in Crop Acres			Percent Change in Crop Prices
Corn	-4.7	3.0	-3.1	13.1
Sorghum	2.5	10.5	3.5	11.4
Barley	-1.9	2.9	-0.1	4.7
Oats	-5.8	0.0	-4.6	21.3
Wheat	-3.8	1.0	-3.2	3.3
Rice	-19.3	6.7	-8.4	6.4
Soybeans	-7.7	2.5	-5.8	10.3
Cotton	-3.2	1.9	-1.2	4.8
Silage	-8.4	-8.3	-8.2	2.2
Hay	-12.0	-0.8	-7.8	2.3
Total	**-6.0**	**1.0**	**-4.3**	**NA**
	Percent Change in Returns[1]			
Crop Enterprises	7.7	11.4	9.0	
Livestock Enterprises	-4.5	-1.9	-2.4	
Total Farm Net Cash Return	**5.8**	**4.0**	**4.9**	
	Percent Change in Nutrient and Sediment Losses to Field Boundaries			
N Loss	-5.4	2.0	-3.8	
P Loss	-6.0	1.6	-4.2	
Erosion	-4.5	1.2	-3.1	
N Trapping[2]	-27.9	NA	NA	

[1]Returns for the Mississippi Basin are for the Corn Belt, Northern Plains, and Delta regions because USMP return information cannot be disaggregated lower than USDA farm production regions.
[2]Based on 4 g N/m^2/year from Mitsch et al. 1999.

Adjustments within the basin also result in decreases in phosphorus losses and soil erosion. Each of these reductions also generates water quality benefits. However, the increases in crop prices result in more intensive production on existing acreage outside the basin, and in an increase in acreage in production. Without any environmental constraints, soil erosion increases in the rest of the U.S., to the probable detriment of water quality. The important thing to note is that, under the assumptions about N-loss reduction per m^2 of buffer and the cost of retiring land from production and restoring vegetation, this massive riparian buffer restoration effort is not cost-effective at reducing N losses relative to fertilizer reductions or wetland restoration (Table 4.7).

TABLE 4.20. Performance of a 109,266-km^2 ((27-million-acre) riparian buffer restoration program.

Performance Indicators	Enrolled Proportional to N Yield	
	Total (millions of $)	**$/Hectare[1] ($/Acre)**
Annual Direct Costs	2,157	197 (80)
Easement	1,645	151 (61)
Restoration	512	47 (19)
Annual Indirect Costs	**35,004**	**3,203 (1,296)**
Farm Cash Income[2]	-3,634	-333 (-135)
Consumer Surplus	21,136	1,934 (783)
Net Social Benefit	17,502	1,602 (648)
	Metric Tons	***Kg***
N Reductions[2]	**691,983**	**63.3**
Field Losses[3]	254,918	23.3
Trapping[4]	437,064	40.0
P Reductions	**907**	**0.1**
Sediment Reduction	**51,013,719**	**4,668.8**

[1]*Per acre of riparian buffer.*
[2]*Field losses and trapping not strictly additive.*
[3]*Estimated losses to the edge of the field or the bottom of the root zone are based on EPIC simulations.*
[4]*Based on 4 g N/m^2/year from* Mitsch et al. 1999.
Source: U.S. Mathematical Programming Model analysis, Economic Research Service, USDA.

4.5 MIXED POLICY

The apparent economic sensitivity of agriculture in the Mississippi Basin to nitrogen-use restrictions suggests that meaningful nitrogen-loss targets will be difficult to accomplish solely by restricting nutrient input. One alternative policy is to combine moderate nitrogen-use restrictions with technologies that slow or prevent the loss of nitrogen from the fields. Restored and constructed wetlands appear to be a promising "natural" technology that can sequester nitrogen and reduce the fraction that is eventually discharged into the Gulf of Mexico. One of the conclusions from the literature is that a mixed policy that uses several different tools may be the most cost-effective approach when there is a great deal of variation in physical and economic conditions (Shortle and Abler 1994).

We explored how a policy combining wetland creation and fertilizer reduction compared with a wetland- or fertilizer-only policy. We reviewed these results to identify which combination of fertilizer reduction and wetland creation might achieve the 20% N-loss goal at least cost. We selected the proportional 5-million-acre wetland restoration with a 20% reduction in N fertilizer use. When running the USMP model, we first removed the 5 million acres from the cropland base (replicating the proportional 5-million-acre wetland scenario described above), and then applied the basin-wide 20% fertilizer reduction constraint to the remaining cropland. The results are summarized in Tables 4.8–4.10. The sum of the N reductions for the individual policies is 18.8%, assuming high wetland N filtering, which is slightly less than the 20% N reduction goal. Price and acreage impacts are substantially lower than for the fertilizer-based policies that achieve a similar goal. Social welfare is reduced by 0.5%, or about $4.8 billion, which is substantially less than the wetland-only policy (Table 4.7). Net returns to crop producers increase in both the basin (represented by the Corn Belt, Northern Plains, and Delta) and the U.S., and export volumes decrease for all crops. Nitrogen loss increases slightly in the rest of the U.S., and soil erosion increases slightly in the basin and in the rest of the U.S.

4.6 SUMMARY

Figure 4.6 summarizes the results of the fertilizer use and wetland restoration scenarios. Results are compared on the basis of cost per unit of N-loss reduction (ignoring benefits for the time being). Costs are defined as changes in consumer-producer surplus for the fertilizer strategies, and consumer-producer surplus plus wetland easement and restoration costs for the wetland and buffer strategies. Note the parametric N-loss curve, which represents the least cost for achieving N-loss reductions.

The strategy that comes closest to the least-cost solution is the 45% fertilizer restriction. The 20% fertilizer restriction costs less, but does not meet the policy goal. The wetland and buffer strategies are all much more costly, due mostly to the opportunity cost of retiring cropland and restoration costs. The combined policy does somewhat better, but is still more expensive than the fertilizer strategy because of the land retirement opportunity costs and wetland restoration costs.

4.7 POINT-SOURCE REDUCTION THROUGH NONPOINT TRADING

Agriculture is only one source of nitrogen in the basin, albeit the largest. An alternative to addressing agricultural sources is to address point sources, which are much easier to observe and to control. For this analysis, we assume a policy of requiring all point sources discharging N in the Mississippi Basin to install advanced nutrient removal technology. Such a policy would be the easiest to implement given current water quality laws, where only point sources of pollution are controlled through command-and-control policies. We impose what we think to be a fairly stringent requirement that all treatment plants achieve a discharge level of 3 mg/l.

An alternative to reducing only point sources is to reduce nonpoint sources, either in combination with point sources or alone. If the cost of reducing a unit of N is less for nonpoint sources, then efficiency considerations would suggest that reductions are targeted to these sectors first. A way to do this under current water quality laws is through a trading system. Simply, trading allows point sources to “purchase” required reductions from cheaper sources as a means of meeting their discharge requirements.

For example, suppose a point source is required to reduce its N loadings by 50%, and that this costs the firm $150 per pound of N reduced. Also suppose that improved nutrient management practices on cropland in the same basin could produce the same reduction in N loads for only $20 per pound. Efficiency is gained if the point source can pay the nonpoint source to install nutrient management practices, rather than installing the more expensive treatment technology. An equivalent reduction in N discharge can be achieved at a lower cost to society. Such a system is being used in some basins in North Carolina (EPA 1992). This can be considered a privately funded cost-share program for nutrient management. For the purposes of this analysis, we assume there is no complementary, publicly supported program or policy for reducing agricultural nonpoint-source pollution.

Information on the costs of different methods for modifying existing municipal sewerage treatment plants so as to further reduce N was adapted from cost equations developed originally by Hazen and Sawyer and Smith Associates (1988), as modified and reported in Camacho (1992) for the Chesapeake Bay Program. The retrofit planning cost curves provide estimates for four types of secondary N treatment to accomplish biological N removal: extended aeration, activated sludge, activated sludge with nitrification, and activated sludge with fixed film. Curves were estimated for two levels of total nitrogen (TN) removal (8.0 mg/l and 3.0 mg/l seasonal), for two levels of plant discharge size (0.5–5.0 mgd and 5.0–30.0 mgd), and for areas with and without bans on phosphate discharges. Equations for annualized capital and operation and maintenance costs, in 1990 constant dollars, were estimated as nonlinear equations of the form:

$$\text{Capital} = a\,(\text{Flow})^{b}$$

and

$$\text{O\&M} = c\,(\text{Flow})^{d}$$

where

Capital = capital costs
O&M = operation and maintenance costs
Flow = design flow in millions of gallons per day (mgd)
a,b,c,d = regression coefficients and exponents

These cost curves generally drop rapidly from negligible discharges to 5 mgd, then remain relatively flat but decreasing over the 5–30 mgd range (see Figure 4.7). For this report, we assume that all municipal sewage treatment plants must install activated sludge with nitrification, so that an N-discharge target of 3 mg/l is met. This particular technology offered the best combination of performance and cost over the range of treatment level required in the basin.

Data on nitrogen discharged from treatment plants and total discharge were obtained from each county in the basin (Gianessi and Peskin 1984). The cost equations were evaluated for a hypothetical plant, with flow and N discharge equal to the total municipal and industrial discharge in each county in the basin. The total capital and O&M cost of the retrofit was then divided by the difference in N discharge between the base condition and the 3.0 mg/l level, to calculate an annual cost per pound of N reduction, which ranged from $1.79 to $22,976.81.

The supply of N credits from agriculture was estimated for each region within the USMP agricultural model. The model is used to estimate price and quantity effects on the agricultural sector in response to the creation of a market for N reductions.

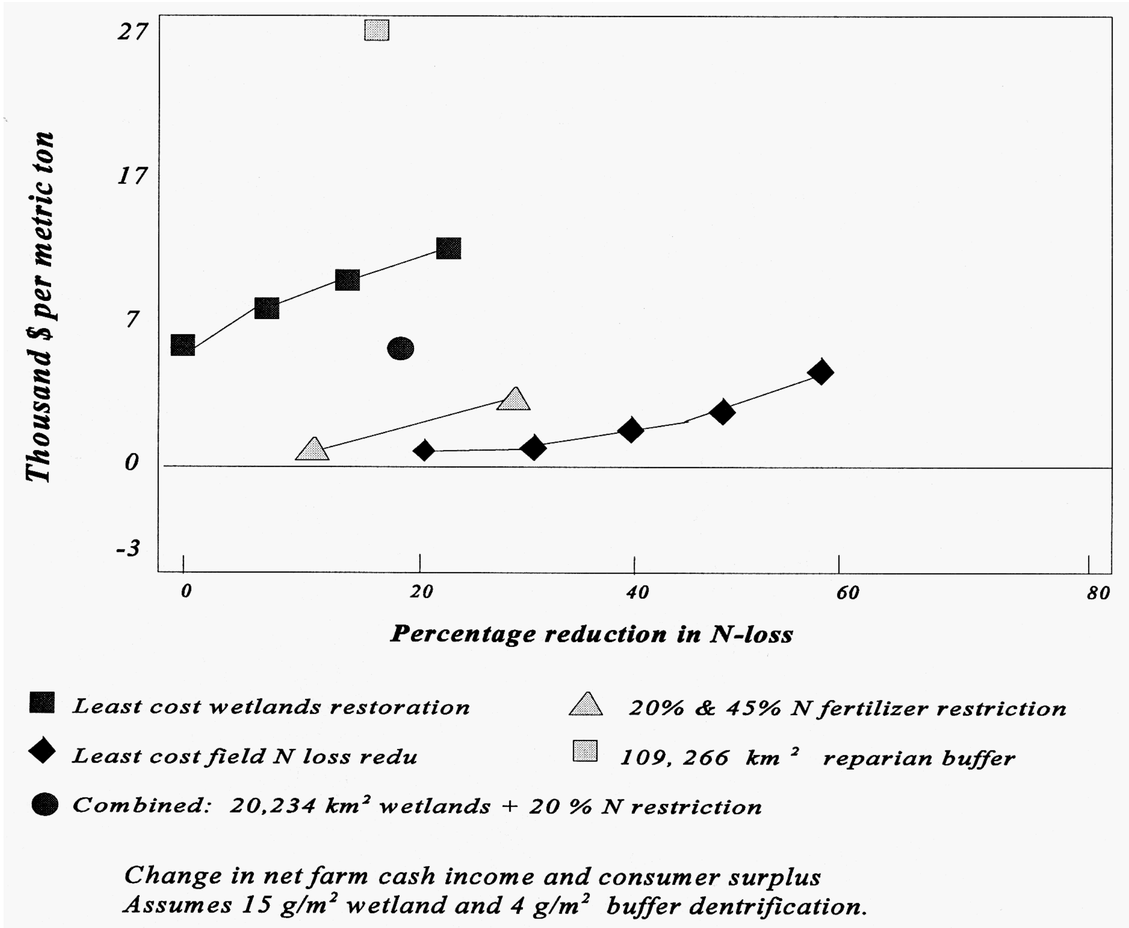

FIGURE 4.7. Social cost per unit of N reduction, without intra-basin benefits, by control method.

To represent the demand for nonpoint reductions in N on the part of point sources within the USMP model, meta-cost functions were estimated for each model subregion from the estimated county wastewater treatment costs (Figure 4.8). These meta-cost functions can be viewed as demand functions for trading point- and nonpoint-source N reductions (Figure 4.9). That is, the curve shows the cost point sources in the subregion need to incur in order to achieve a given cumulative reduction in N discharge. Point sources should be indifferent to paying that cost for N reduction by retrofitting or by compensating farmers for nonpoint-source reductions in N of equivalent size. Average N effluent concentrations, and average treatment costs weighted by the relative N discharge in each county, are shown for each subregion in Table 4.21.

Given the demand for nitrogen reductions, the agricultural sector in each region can supply N-reduction credits by changing fertilizer application rates, switching production practices, or growing different crops. The amount of credits sold within a region depends on the demand for N credits and the costs of reducing N losses in agriculture. Agriculture will supply N credits up to the point where the marginal cost of producing the next credit is greater than the marginal cost of treatment, or until the total point-source demand is met. We used the USMP model to determine the amount of N reduction purchased by point sources in each region.

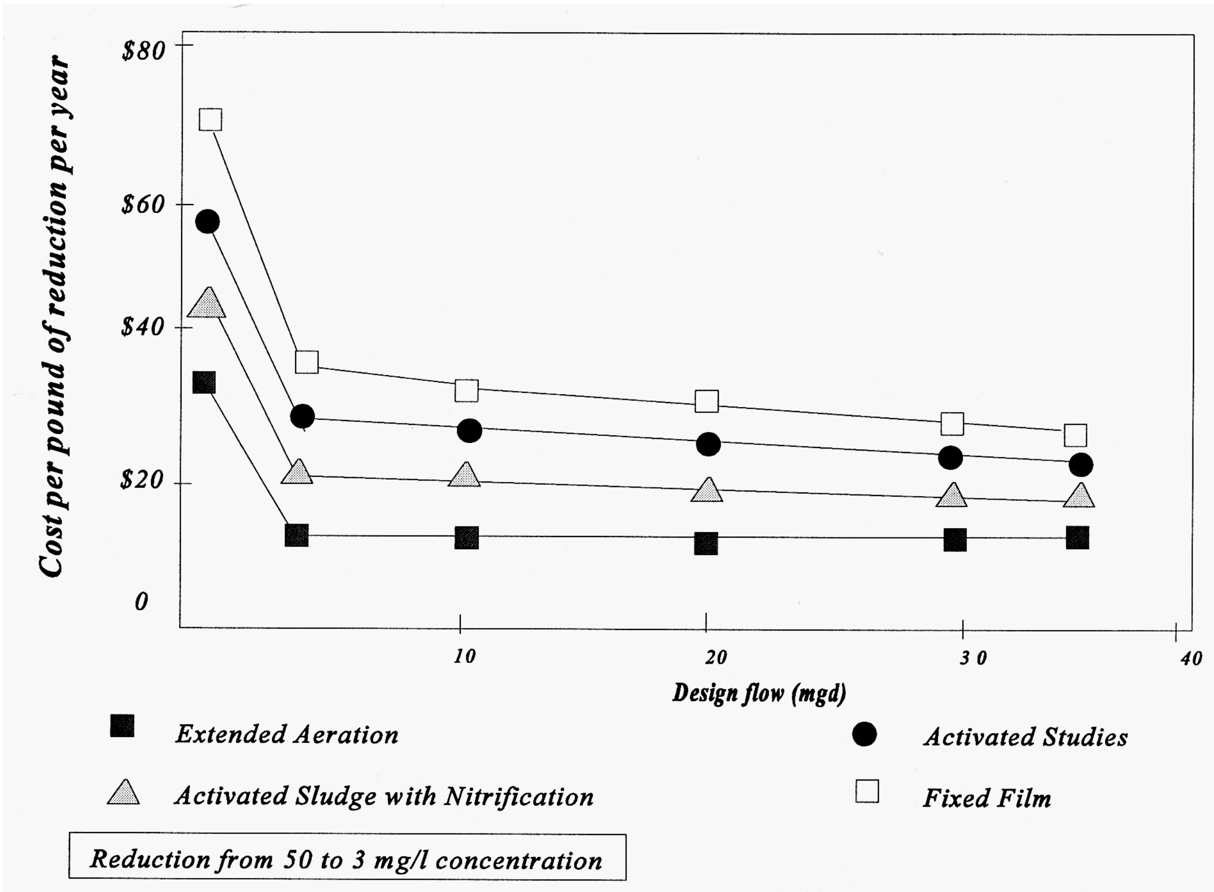

FIGURE 4.8. Planning-level biological nitrogen removal retrofit cost curves.

The price of the marginal sale of N credits in each region is reported in Table 4.21, as well as the amount of N credits purchased by the point sources. The results are reported for each of the 21 USMP regions that are in the Gulf of Mexico drainage area. Given the opportunity to purchase N-reduction credits from agriculture within their respective regions, point sources would pay agriculture to reduce N loads by 407.93 million pounds, or 47% of total point-source reductions obtainable by the required technology. In 12 regions, point sources could meet their total responsibility by buying credits. In the other nine, agriculture could not meet the entire demand because point sources could meet at least part of their obligation more cheaply by installing the advanced treatment technology. However, credits were purchased in all regions. Point sources realized cost savings of about $14 billion by not having to install advanced treatment and instead purchasing N reductions from agriculture.

The ability of farmers in the Mississippi Basin to sell N-reduction credits to point sources has important implications for agriculture in the basin and the rest of the country. Table 4.22 summarizes the changes in crop prices, acres planted, and farm income in the Mississippi drainage area and rest of the U.S. With the exception of hay and silage, the prices of most commodities increase. Within the basin corn, soybean, wheat, and rice are shifted into other crops, primarily oats, silage, and hay. Elsewhere in the U.S., the acreage planted to all crops, except soybeans and hay, increases. Total acreage planted increases in the basin and decreases in the rest of the U.S.

TABLE 4.21. Summary of point-source control costs, total demand for credits, amounts purchased, and the cost of a marginal credit.

Region	Effluent Concentration *mg/l*	Total Point-Source N Reduction Requirement *millions of lbs*	Weighted Average Treatment Cost *$/pound*	Pounds N Traded *millions of lbs*	Cost of Marginal Credit *$/pound*
NTN	14.8	45.21	40.16	1.27	98.83
LAF	28.1	1.78	35.97	1.78	20.75
LAK	14.6	41.14	40.69	41.14	15.67
LAM	22.2	97.41	20.62	97.41	11.55
CBM	19.7	250.41	29.85	135.21	16.62
CBN	21.1	48.36	36.42	10.55	45.99
CBO	30.8	1.50	48.94	1.50	35.12
NPF	30.5	5.84	48.17	5.84	31.18
NPG	24.8	3.49	64.64	3.49	44.08
NPH	29.6	15.20	39.44	15.20	15.20
NPM	27.8	27.37	24.59	27.37	18.34
APN	18.8	94.02	34.17	1.29	187.97
STN	27.3	29.99	34.92	3.16	61.46
DLN	25.9	11.43	29.80	3.36	31.66
DLO	25.6	76.37	24.04	8.38	41.92
SPH	18.6	9.40	41.64	9.40	25.46
SPJ	28.0	65.86	24.29	4.69	53.31
SPM	22.8	6.74	29.82	6.23	20.56
MNF	25.0	1.57	53.03	1.57	26.69
MNG	21.8	27.68	30.26	27.68	21.94
MNH	23.4	1.41	61.69	1.41	38.14
Total		**862.18**		**407.93**	

Note: The locations of the regions appear in Figure 4.

TABLE 4.22. Changes in crop prices and acreage planted in trading scenario, by Mississippi drainage and rest of U.S.

Crop	Percent Change in Price	Percent Change in Acreage Planted	
		Miss. Basin	Rest of U.S.
Corn	2.4	-0.3	0.6
Sorghum	1.6	1.3	0.0
Barley	0.4	0.0	0.0
Oats	4.2	6.4	0.0
Wheat	0.5	-0.3	5.4
Rice	3.0	-5.3	6.7
Soybeans	0.9	-0.7	-0.8
Cotton	0.6	0.6	1.9
Silage	-0.1	2.5	0.0
Hay	-1.5	8.0	-0.8
Total		**1.1**	**-0.1**

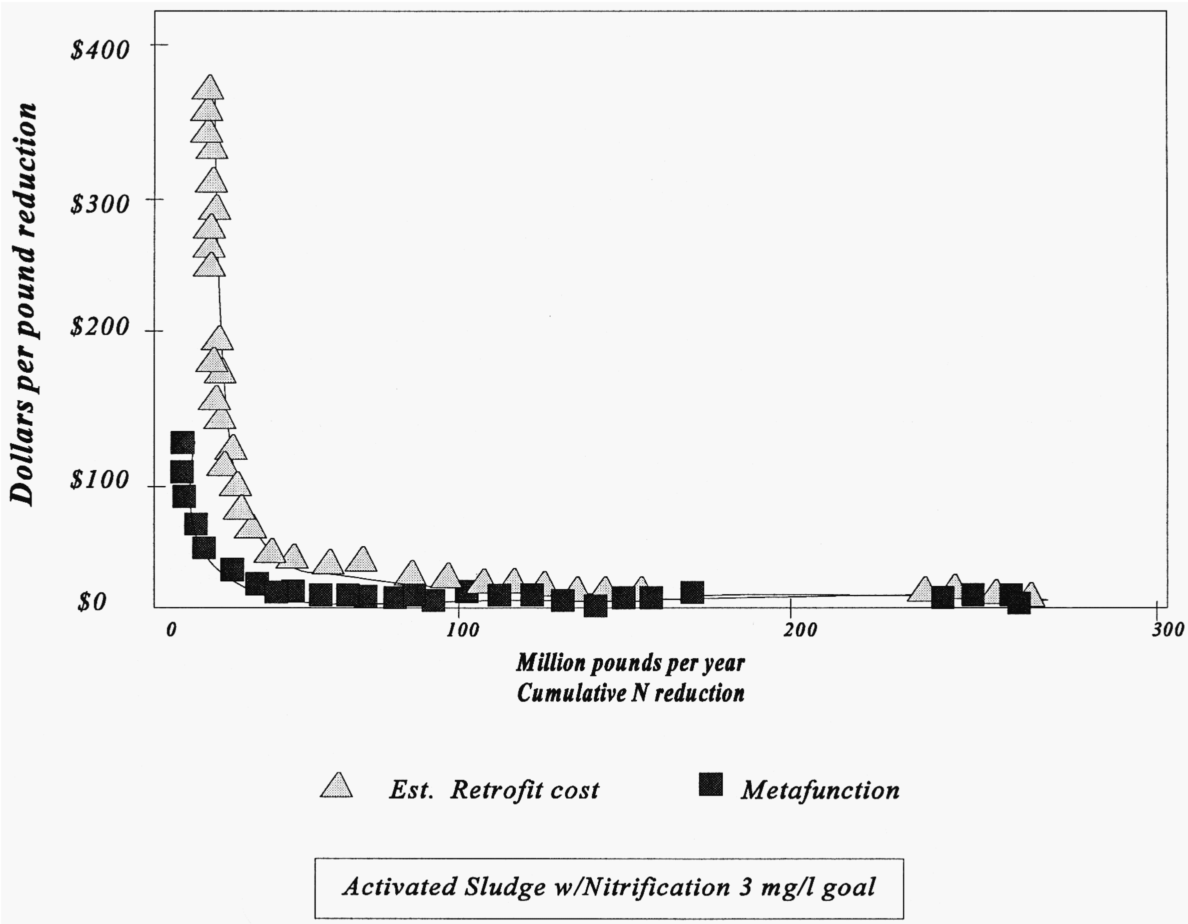

FIGURE 4.9. Demand for N reduction for point-source trading.

This increase may be an indication of a moral hazard problem. The newly created market for N-loss reductions encourages the expansion of cropland so that farmers have more N-reduction credits to sell. Total acreage planted in the U.S. increases by about 0.8%, and net cash returns for crop production in the entire U.S. increase by about 1% ($465 million).

The changes in crop production have implications for environmental quality (Table 4.23). In the basin, N losses are reduced, as might be expected. However, these N-loss reductions are nowhere near the 20% goal stated previously. This demonstrates point sources' relatively small contribution to the overall nitrogen loading problem. In the rest of the country, N losses increase as the rise in the prices of some major crops spurs growth in production and fertilizer use. Changes in production practices in the basin result in a small increase in soil erosion, primarily from an expansion of moldboard plowing. The increase in erosion could have negative consequences for water quality in the basin. Changes in crops and management practices also result in greater phosphorus losses in the basin, and virtually no change in phosphorus losses outside the basin. The consequences of changes in erosion and phosphorus losses would need to be considered in a complete benefit–cost assessment of a trading program.

TABLE 4.23. Changes in physical environmental indicators for trading scenario, by Mississippi drainage and rest of U.S.

Indicator	Percent Change	
	Miss. Basin	Rest of U.S.
Nitrogen Loss	-0.8	0.2
Phosphate Loss	0.5	0.0
Soil Erosion	0.9	0.0

CHAPTER 5

Environmental Benefits

As with costs, the range of potential benefits can be extensive. Costs are often amenable to being calculated on an accounting or engineering basis, and a model like USMP makes this process appear even more precise than it is likely to be in fact. Benefits, especially when related to environmental issues, are often less defined because the public values things that have not been traditionally thought of in terms of dollars and cents.

In this case we tried to look at a broad range of benefits corresponding to our broad range of costs. Some benefits should be amenable to quantification in economic terms, like benefits to commercial fisheries in the Gulf of Mexico. Others, like the restoration of ecosystem health in the Gulf, are important, but we do not feel comfortable placing a price tag on them. While the restoration of these ecological communities may be nearly impossible to value in a direct monetary sense, they can be reasonably expected to provide some basis of support for the species that comprise the Gulf's commercial and recreational fishing industries.

Improved water quality within the Mississippi Basin certainly has some economic value, as evidenced by studies from specific watersheds or relating to specific water uses. While we reference examples of these studies, we are unable to aggregate and extend them to estimate the basin-wide benefits of improved water quality. We can give estimates for reduced erosion damage for the basin, which can be counted as a benefit when farm practices adopted to reduce nitrogen losses also reduce erosion. Finally, to some extent because we have national programs to restore wetlands, there is a body of literature estimating benefits from wetlands, and we can estimate the within-basin benefits from creating wetlands to reduce nitrogen flows to the Gulf.

5.1 SUMMARY OF GULF OF MEXICO BENEFITS

Because a number of marine species in the northern Gulf of Mexico are targets of commercial and recreational fishing, they have direct economic value that can, in principle, be estimated. Benefits would be expected to arise from reduced hypoxia if, under current conditions, the revenue and/or cost structures of the commercial industries are being negatively affected by the presence of hypoxia. Such negative effects may have occurred due to decreases in the stock of commercially viable species, shifts in the location of fishing grounds associated with those species, or changes in the quality of the species being harvested. These changes, if they occurred, would have led to changes in the target species being harvested, perhaps to species less preferred by consumers and, thus, bringing a lower dockside price. Shifts in the fishing grounds could have led to increased effort required for a unit harvest, although it is also possible that effort could be reduced if hypoxia crowded species into nearshore waters.

Estimating these benefits in the commercial fishery may be particularly straightforward, as the information needed to assess hypoxic effects is generally detailed and systematically collected for commercially important species. While the economic importance of the recreational fishery both in the Gulf of Mexico and in the freshwater riverine systems of the Mississippi Basin cannot be overstated, little systematic data have been collected that would allow the identification of the economic effects of hypoxia. This lack of information is especially acute with respect to the charter and trip-support industries.

The Topic 2 report, *Ecological and Economic Consequences of Hypoxia*, estimated the benefits resulting from reduced hypoxia in the Gulf of Mexico. Table 5.1 briefly summarizes their conclusions (Diaz. and Solow 1999). The implicit baseline used in the Topic 2 analysis was current conditions in the ecological communities and the commercial fishery. Historical time-series and geographical data were examined to discern the effects of hypoxia on the Gulf of Mexico ecosystem and the fishing industry over the last 30 years. Because of conceptual and measurement problems, we could not calculate reliable estimates of the noncommercial use and nonuse values of restored ecological communities. Thus, the following discussion focuses on the other potential benefits of reduced hypoxia in the Gulf—namely, the impacts on the commercial and recreational fisheries.

TABLE 5.1. Summary of estimated benefits associated with hypoxia reduction in the Gulf of Mexico, as presented in the Topic 2 report.

Potential Benefits of Hypoxia Reduction	Conclusions from Topic 2 Report[1]
Restoration of ecological communities in the Gulf of Mexico.	Data not available to estimate benefits.
Increased commercial harvesting of white and brown shrimp.	Given available data, no estimable benefits from hypoxia reduction.
Increased commercial harvesting of Gulf menhaden.	Given available data, no estimable benefits from hypoxia reduction.
Increased commercial harvesting of red snapper.	Given available data, no estimable benefits from hypoxia reduction.
Increased recreational harvesting.	Data not available to estimate benefits.

Source: Diaz and Solow 1999.

The Topic 2 analysis focused primarily on the two main shrimp species in the northern Gulf: brown shrimp and white shrimp. The reasons for concentrating the search for benefits on these two species were twofold. First, these species are commercially important. As a result, the data for these fisheries are relatively good, particularly in terms of their geographic resolution and potential overlap with areas of hypoxia. Second, as benthic species, their potential for experiencing hypoxic effects would likely be relatively high, although a search of the literature identified only one quantitative assessment of the effects of hypoxia on a commercial shrimp fishery, and those effects were considered insignificant (Zimmerman et al. 1996).

The Topic 2 assessment of the economic benefits associated with hypoxia reduction suggested that, given the available data, no effect in the shrimp fisheries data could be attributed with high confidence to hypoxia. It was noted that the failure to identify hypoxic effects in the fisheries data was consistent with the results of the broader Topic 2 ecological study. Given that no historical effect could be identified, the implication is that efforts to reduce hypoxia in the Gulf of Mexico will not generate benefits for the white and brown shrimp fisheries.

Additional analyses were performed for two other species—Gulf menhaden and red snapper—to address specific concerns about the affects of hypoxia and potential hypoxia reductions on the spawning and grow-out waters of these species. No statistically reliable impacts that they could attribute to hypoxia were identified in the historical data. The inability to associate hypoxia with significant changes in the historical data for the important northern Gulf of Mexico fisheries led the Topic 2 report to conclude that there was no need to assess the direct economic benefits of reducing hypoxia in the Gulf of Mexico. Given the lack of available data, no attempts were made to estimate the benefits that might accrue to the recreational fishing industry, although there is no reason to assume that the results would be significantly different from those generated for the commercial industry.

It is worth emphasizing, as did the Topic 2 report, that failure to identify hypoxic effects in the commercial fisheries data does not mean that they do not exist. But, if hypoxic effects do exist, their magnitude must be small in relation to other sources of variability in the data. The ability of the Topic 2 team to account for different sources of variability in the fisheries data was limited by the quality and extent of the information. In particular, data about the severity of hypoxic conditions in the Gulf of Mexico were limited to estimates for 1985–95. Given the naturally high variability of fisheries data, this time series may have been too short to establish a relationship between the severity of hypoxia and reactions within the commercial fisheries. The Topic 2 team did attempt to develop a proxy variable for hypoxia based on nutrient discharge, thereby extending the length of the time series, but no linkage with the commercial fisheries was discovered. Still, the inability to identify hypoxic effects in the historical data does not imply that effects would not occur should hypoxia continue or expand in the Gulf of Mexico, as experience with hypoxia in other geographic areas indicates that effects of hypoxia become progressively larger as hypoxia worsens (Caddy 1993).

5.2 INTRA-BASIN BENEFITS

While hypoxia reduction in the Gulf of Mexico may have many potential direct economic benefits, many indirect benefits may also be generated within the Mississippi Basin as a result of alternative nutrient management policies. Reducing nutrient loads by more efficient management of organic and inorganic fertilizers in crop production and concentrated livestock operations can generate monetary benefits for agricultural producers by lowering nutrient input or disposal costs. Alternative nutrient management strategies can also reduce soil erosion, depending on the adopted tillage technologies. Given the basin's size, it is unlikely that a complete information set exists to directly estimate these agricultural benefits. However, estimates can be obtained by using established agricultural policy simulation models and information contained in the literature.

Along with the potential indirect agricultural benefits from reduced hypoxia, reduced nutrient loadings from agriculture should decrease contamination of surface- and aquifer-based drinking-water supplies, and thus reduce the cost of obtaining potable drinking water. Decreased contamination of surface waters could also significantly expand the area of recreational waters suitable for such nonconsumptive uses as swimming—an important objective of the Clean Water Action Plan. Reduced nutrient loads—especially those originating from livestock operations—should decrease the potential for the development of various waterborne human pathogens. Using management technologies, such as constructed and restored wetlands, to reduce the downstream flow of nutrients would generate both consumptive (hunting, fishing) and nonconsumptive (bird watching, boating) benefits. Although all of these indirect benefits are potentially quantifiable, at best only partial information sets exist upon which to base reliable monetary estimates.

Loss of nutrients, sediment, and pesticides from agricultural land may cause economic losses for landowners and others effected by reduced environmental quality. Costs to landowners may derive from reduced farm productivity due to soil and nutrient loss or from expenditures to correct or avoid problems, such as clogged drainage ditches, flooding, or water contamination. These "on-farm" types of impacts may be experienced by farm operators and/or by adjacent landowners.

Nutrient, sediment, and pesticide losses from agricultural land may also reduce the quality of freshwater resources and related ecosystems, and thereby reduce the value of services provided to their users. These services may include recreational uses, such as fishing and swimming; consumptive uses, such as drinking water; and aesthetic values associated with proximity to the resource. Other values associated with preservation of wildlife habitat and ecosystem diversity may also be important. While the on-farm impacts may be substantial, the off-farm benefits of control are generally considered to be much more significant (Crosson 1986).

Economic measures to determine the magnitude of these losses vary depending on the type of activity. Conceptually, any measure seeks to estimate the "willingness to pay" to avoid a loss or, possibly, to benefit from an improvement in the resource. Willingness to pay has been accepted as the standard measure of changes in economic welfare for many federal environmental management evaluations (U.S. Water Resources Council 1983; Morgenstern 1997). In the case of on-farm impacts of nutrient, sediment, and pesticide losses, direct expenditures to correct or avoid problems are one measure of willingness to pay. However, losses to users of water resources are often more difficult to identify and may involve both use and nonuse values.

Use values are related to recreational or other consumptive activities and are measured through revealed or stated preference methods of economic valuation (Freeman 1993). Nonuse values are less tangible and may result from a variety of concerns about water resource preservation; these values are typically measured through stated preference methods. Many studies of nonpoint pollution provide estimates of "damages" as a measure of the potential economic benefits from reduced loadings. These estimated damages may understate the total benefits if they do not include the full range of potential benefits (Raucher et al. 1991).

This section discusses in greater detail the potential river system and watershed benefits from strategies for reducing nitrogen loadings, and provides empirical estimates of these benefits within the Mississippi River drainage basin, when available. Due to the limited research on the benefits of reduced loading strategies, no aggregate estimate is provided for the drainage basin. Also, these benefit estimates are not directly related to particular control strategies.

Reductions in nutrient applications to crops may have very different effects on agricultural pollutant loadings to streams and rivers than other control strategies, such as changes in tillage practices. Due to the lack of information about the linkage between control strategies and economic benefits to the watershed, no attempt is made to identify benefits with any particular strategy. Also, the choice of control strategy may trigger other pollutant concerns, such as on-farm ground-water contamination with increased nutrient retention (USDA 1995c; Crutchfield et al. 1997).

5.2.1 Sediment- and Erosion-Reduction Benefits

5.2.1.1 ON-FARM BENEFITS

The on-farm economic costs of erosion occur from losses in crop yield and/or increased costs of practices to mitigate the effects of erosion. Soil erosion is the movement of soil by water and wind from one place to another. Soil moved by erosion carries nutrients and organic matter from the field and makes the crop-rooting zone more shallow. This may lower the soil's productivity by reducing its fertility, restricting its ability to infiltrate and hold water, and making the soil more difficult to till. On-farm costs of erosion are measured by long-term reductions in potential yields and increased costs of maintaining soil fertility through fertilization (Crosson 1998).

The USMP model uses EPIC to estimate long-term soil productivity costs from soil erosion. The nutrient management scenarios generally reduced soil erosion in the Mississippi Basin and increased it elsewhere. On average, the alternative nitrogen-reduction strategies reduced productivity costs for the U.S., even though erosion generally increased. The pattern of erosion reductions and increases was such that overall long-term productivity increased. Productivity benefits are summarized in Table 5.2.

TABLE 5.2. Summary of productivity and water quality benefits from reduced soil erosion.

Scenario	Productivity (Millions of Dollars)	Water Quality (Millions of Dollars)
N-Loss Reduction		
20%	10.2	67.8
30%	14.5	121.0
40%	19.8	205.2
50%	30.0	269.5
60%	33.5	358.8
Fertilizer Reduction		
20%	1.7	10.2
45%	6.9	31.8
500% Fertilizer Tax	6.9	31.8
Wetland Acreage[1, 2]		
1,000,000	0.8	2.3
5,000,000	2.8	12.2
10,000,000	4.6	23.6
18,000,000	8.9	46.8
Buffer[3]	10.5	51.3
20% Fertilizer Reduction and 5-Million-Acre Wetland	4.9	20.6

[1]Social welfare for wetland strategies includes changes in consumer and producer surpluses, plus wetland restoration costs.
[2]Wetland filtering capacity: 15 g/m^2.
[3]Buffer filtering capacity: 4 g/m^2.

5.2.1.2 OFF-FARM BENEFITS

The off-farm benefits of nutrient- and sediment-loss reductions cover a broad array of water-related uses and activities. The easiest types of benefits to quantify are those that occur when nutrients and sediments impair direct uses of water in a river or lake. These direct uses may include in-stream activities, such as recreational and commercial fisheries, navigation, and/or other recreation, such as swimming or picnicking adjacent to the water body. Other direct uses include the diversion of water for domestic or industrial purposes, such as drinking water, manufacturing, or power production. Benefits can result from reduced treatment costs, reduced health risks, or reduced wear and tear on equipment.

Due to the broadly dispersed nature of damages from nonpoint-source emissions, relatively few studies of potential benefits from nonpoint-source controls have been undertaken. One of the most comprehensive and relevant for the Mississippi River drainage is Ribaudo's (1989) regional breakdown and extension of Clark et al.'s (1985) national estimates of erosion damages. Table 5.3 presents updated estimates of off-farm damages for various direct uses and activities, by region. While the estimates cannot be decomposed into damages related to nutrients versus sediments, they provide useful guidance on the magnitude of possible benefits.

TABLE 5.3. Off-site damages from soil erosion, by farm production region.

Region	Damage[1] per Ton of Erosion 1998 $
Appalachian	1.99
Corn Belt	1.62
Delta	3.44
Lake States	5.27
Mountain	1.58
Northeast	9.95
Northern Plains	0.80
Pacific	3.49
Southeast	2.70
Southern Plains	2.85

[1]Damages include: freshwater fishing, flooding, water storage, marine recreation, commercial fishing, navigation, roadside ditches, municipal water treatment, municipal and industrial water use, and steam power cooling.

Source: Ribaudo 1989.

Although other studies exist of the estimated contribution of specific rivers to regional economic activity within the basin (e.g., Vicory and Stevenson 1995), only a few conducted within the Mississippi River drainage basin provide estimates of the possible river system and watershed benefits from agricultural emission controls. For example, Forster et al. (1987) evaluated water treatment costs for 12 municipal water facilities in western Ohio. Their statistical analysis indicated that upstream erosion increased water treatment costs. They concluded each 10% reduction in annual gross soil erosion would reduce treatment costs by 4%, resulting in significant savings within Ohio. Other possible benefits to recreation, navigation, etc., were not considered.

The USMP estimates off-site damages from soil erosion and associated runoff using Ribaudo's estimates (1989). All the nitrogen-reduction strategies resulted in overall water quality benefits from reduced soil erosion. While damages may have increased in regions experiencing increased erosion, greater benefits are obtained in regions where soil erosion decreases. Results are summarized in Table 5.2.

5.2.2 Wetland Benefits

5.2.2.1 WETLAND VALUATION STUDIES

Public recognition of the value of wetland functions has grown rapidly over the past 25 years. However, the wetland values from such outputs as food and recreation, from such indirect uses as water quality and flood control, from future direct outputs or indirect uses (biodiversity or conserved habitats), and from the "existence" value of these wetlands often cannot be captured by individual landowners in the marketplace. Many now-recognized wetland benefits are nonmarketed goods, such as water quality and wildlife preservation. While these wetland functions are important to society, they have often been undervalued relative to conversion of wetlands to other land uses.

Although some values derived from wetlands can be determined by using market transactions or by using income attributable to each factor of production used to produce marketable commodities (e.g., Lynne et al. 1981; Batie and Wilson 1979; Bell 1989), most economic values associated with wetland benefits must be estimated using nonmarket techniques. Eliciting use values with nonmarket techniques involves either revealed-preference approaches, such as travel-cost or hedonic methods, or expressed-preference approaches, such as contingent valuation and conjoint analysis (Scodari 1997; Anderson and Rockel 1991; Braden and Kolstad 1991; Freeman 1979). Values of people who do not use wetlands reflect the importance of the continued existence of the resource, or the option of using the resource in the future.

Travel-cost methods are used for recreation sites, where the assumption is made that the cost of traveling to the site and foregoing income from working to use it are revealed measures of the value users place on the resource (Clawson and Knetsch 1966). Hedonic methods decompose the observed values of such goods as housing into various attributes, including environmental amenities that may influence price (Farber 1987). Contingent valuation directly elicits values through surveys, and can be used for both use and nonuse values (Bergstrom et al. 1990; Loomis et al. 1990). Finally, ecological functions provided by wetlands can be valued using replacement- or avoided-cost methods that price the service provided in terms of equivalent man-made services (e.g., nutrient filtering), or in terms of avoided damages (e.g., from flooding or hurricanes) (Farber 1987).

Marketed goods supported by wetlands include fish and shellfish production supported by nursery and feeding areas, and fur trapping supported by habitat for fur-bearing animals. Eight studies in Florida and Louisiana Gulf waters from 1975 through 1990 estimated values (in 1992 constant dollars) ranging from $7 to $1,259 per acre of wetlands (Table 5.4). Fish and shellfish production generally supports higher values, averaging $624 per acre, while fur trapping supports values averaging $137 per acre. Median values are equal to the means. These values would generally apply only to coastal or estuarine wetlands added to the drainage that directly communicate with brackish or saline waters and cannot be extended to freshwater wetland systems.

TABLE 5.4. Economic values of wetland functions, Mississippi River drainage.

Wetland Function Valued	Number of Studies	Median	Mean	Range of Means
		1992 Constant Dollars per Acre		
Marketed Goods				
Fish/shellfish life support	6	624	624	7–1,259

Fur-bearing animals	2	137	137	13–261
Nonmarketed Goods				
General nonusers	3	14,916	23,538	2,850–52,848
General users	4	438	2,710	105–9,859
Fishing users	4	7,741	11,105	95–28.845
Waterfowl hunting users	2	142	142	128–156
Recreation users	7	327	942	91–4,287
Ecological functions	9	1,946	4,430	1–21,428
Visual amenity	1	573	573	573

Source: Heimlich et al. 1998.

Most wetland values for nonmarketed goods are associated with fish and wildlife habitat. Non-use values generally measure existence or bequest values derived from the utility of knowing such habitats exist and will be available to support ecosystems into the future. Three studies of general nonuse value associated with willingness to pay for marginal improvements or changes in migratory bird habitat and recreation in Nebraska and western Kentucky provide estimates ranging from $2,850 to $52,484 per acre, with a mean value of $23,538 per acre. The median value of the studies is $14,916 per acre. Nonuse values held by individuals are small, but are influenced by the number of people thought to hold them. For wetlands across the entire Mississippi drainage, the number of people holding such values and, hence, the aggregate value are likely to be large. Many people outside the drainage would also hold nonuse values for wetlands added to the drainage, so these estimates may understate the true nonuse value. These values would apply to all wetlands added to the drainage.

Four studies in North Dakota prairie potholes and Louisiana coastal wetlands captured general willingness to pay for marginal changes in wetlands by wetland users. Averaging $2,710, the mean values estimated ranged from $105 to $9,859 per acre. The median value of the studies is $438. More specific use values for fishing were estimated in four Louisiana, Florida, and Michigan studies, with a mean value of $11,105 per acre, ranging from $95 to $28,845 per acre. The median value of the studies is $7,741 per acre. Two studies of waterfowl hunting values in the prairie pothole region of the Northern Plains and Louisiana coastal wetlands averaged $142 per acre. General recreation uses were valued in seven studies in Louisiana, western Kentucky, the Corn Belt, and the prairie pothole region, averaging $942 per acre, with a range from $91 to $4,287 per acre. The median value of the studies is $327 per acre. These values would apply generally to most wetlands added to the Mississippi drainage.

Values derived from either replacement cost or willingness to pay for marginal improvements in ecological services provided by wetlands were estimated in nine studies in Louisiana and the Corn Belt states. Estimated values averaged $4,430 per acre, ranging from $1 to $21,428 per acre. The median value of the studies is $1,946 per acre. Finally, one hedonic valuation study of the addition to value for housing in Minnesota estimated $573 per acre of wetlands in proximity to housing, based on the visual amenity added to the site.

Based on these estimates, a mean value of about $750 per acre of coastal wetlands restored would account for marketed goods, such as fish and shellfish support and fur trapping. More generally, each wetland restored is likely to support nonuse existence and bequest values of $14,900–$23,500 per acre. An alternative method of estimating these values is to multiply the population of the Mississippi drainage by the $22–$50 per respondent recorded in studies estimating these values. Wetland users value each additional acre of wetlands at $330–$1,585 for general recreational uses, $7,700–$11,000 per acre for fishing, and $142 per acre for waterfowl hunting. Ecological services provided by each additional acre of wetlands are valued at $1,900–$4,400.

Assuming that a million acres of restored wetlands would include 100,000 acres in coastal areas, the median and mean estimates from the literature described above yield estimates of within-basin benefits ranging from $25 billion to $40 billion if the more expansive nonuse values per acre are used, and from $11.8 billion to $21 billion if the more conservative per capita method of measuring nonuse values is used (Table 5.5).

TABLE 5.5. Estimated economic value of wetland restoration aside from hypoxia mitigation.

Economic Activity	**Number of Studies**	**Mean**	**Median**	**Estimated Wetlands Affected**	**Total Economic Value**
		1992 Constant $ per Acre		**Acres**	**Millions of 1992 Constant $**
Direct Economic Losses					
Commercial fisheries and fur	8	750	750	100,000	75
Public Goods—Use Values					
General recreation	11	1,585	330	1,000,000	330–1,585
Recreational fishing	4	11,000	7,700	1,000,000	7,700–11,000
Waterfowl hunting	2	142	142	1,000,000	142
Public goods—Nonuse Values					
Nonuser values	3	23,500	14,900	1,000,000	14,900–23,500
Nonuser values alternate estimate[1]	3	50/capita	22/capita	NA	1,673–3,802
Public Goods-—Ecological Values					
Ecological Services	9	4,400	1,900	1,000,000	1,900–4,400
Total					**25,047–40,702**
Alternate Total					**11,820–21,004**

[1]Based on a 1995 population of 76,039,000 for the Mississippi drainage area.
Source: Heimlich et al. 1998.

Aside from nonuse values, benefits for recreational fishing within the basin comprise the largest category of benefits, followed by ecological services. In the absence of more specific information on the geographic distribution and characteristics of wetlands likely to be restored, benefits from larger restoration targets (5–10 million acres) would be estimated as simple multiples of the 1-million-acre estimate. This ignores changes in the marginal values per acre of wetlands as more and more of the basin's wetlands are restored, and is thus a likely overestimate of benefits from larger restoration targets.

CHAPTER 6

Ranking the Strategies

6.1 SUMMARIZING THE BENEFITS AND COSTS OF HYPOXIA REDUCTION

Chapters 4 and 5 presented estimates of the costs and intra-basin benefits of alternative strategies for reducing nitrogen loadings to the Gulf of Mexico. Table 6.1 combines the agricultural sector adjustment costs and the intra-basin benefits to calculate a net per-unit cost for reducing nitrogen loss, and Figure 6.1 depicts these results. Including benefits effectively reduces the cost of wetland and combined strategies relative to fertilizer-only strategies, because strategies that include wetland restoration provide larger benefits within the basin than fertilizer-only strategies.

We need to keep in mind that a critical driving force for agriculture that makes nitrogen reduction difficult is the extremely high value in use of nitrogen. A pound of additional nitrogen, under the right circumstances, can provide a return of tenfold its cost in corn production. This gives an extremely strong economic incentive for applying insurance levels of nitrogen, which producers highly value.

For the strategies analyzed, a strategy that restores 5 million acres of wetlands and applies a 20% nitrogen fertilizer restriction is the closest practicable, cost-effectively approach for meeting the 20% N-loss goal. The 45% nitrogen fertilizer constraint comes close to mixed policy, and if a lower N-filtering capacity is assumed for wetlands, it would be the most cost-effective. The buffer strategy is the least desirable, due to high opportunity costs of retiring land, low nitrogen filtering relative to wetlands, and small environmental benefits. We could not estimate buffers' impacts on reduced sediment in surface waters; these benefits would lower the buffers' unit costs, but probably not to a degree to change the final result.

The combined policy may have some additional desirable features not captured by a simple cost-effectiveness measure. The mixed policy has a smaller impact on prices, and thus results in smaller adjustments inside and outside the basin. Less impact at the extensive margin in particular may be viewed more favorably, depending on which types of land are converted to cropland. The mixed policy we examined resulted in a 1.6% increase in cropland outside the basin, while the fertilizer reduction resulted in a 5.6% increase in cropland inside the basin.

Any policy that includes wetland restoration would also be viewed favorably in light of the national decision of "no net loss" for wetlands. Such policies would in effect be fulfilling two national mandates: reduce the frequency and magnitude of hypoxic conditions in the Gulf, and contribute toward meeting the "no net loss" goal.

TABLE 6.1. Summary of economic impacts.

Scenario	Social Welfare[1,2]	Erosion Benefits[2]	Wetland Benefits[2]	Net Social Benefits	N-Loss Reduction	Unit Cost
	Millions of Dollars				1,000 Metric Tons	$/Metric Ton
N-Loss Reduction						
20%	-831	78		-753	941	800
30%	-2,677	136		-2,541	1,412	1,800
40%	-6,343	225		-6,118	1,882	3,251
50%	-12,239	300		-1,1939	2,352	5,076
60%	-21,109	302		-20,807	2,822	7,373
Fertilizer Reduction						
20%	-347	12		-335	503	666
45%	-2,922	39		-2,883	1,027	2,807
500% Fertilizer Tax	-14,932	39		-14,893	1,027	14,501
Wetland Acreage						
1,000,000	-406	3	550	147	67	-2,192
5,000,000	-3,115	15	2,751	-349	350	997
10,000,000	-7,525	28	5,502	-1,995	713	2,813
18,000,000	-15,506	56	9,904	-5,546	1,300	4,266
Buffer	-18,014	62		-17,952	692	25,942
20% Fert. Reduction and 5-Million-Acre Wetland	-4,854	26	2,751	-2,077	882	2,354

[1]Social welfare for wetland strategies includes changes in consumer and producer surpluses, plus wetland restoration costs.
[2]Net social costs include social welfare, erosion benefits, and wetland benefits.

Our analysis of strategies that included wetland restoration assumed that wetlands would filter 15 grams of N per square meter of wetland. A higher filtering efficiency would reduce the number of acres of restored wetlands necessary to achieve the nutrient-reduction goal, thereby reducing the unit cost. For the 10-million-acre wetland-restoration strategy summarized in Table 6.1 to be as cost-effective as the mixed strategy, wetland filtering efficiency would have to double, to about 30 grams of N per spare meter. Such a level of filtering capacity has been recorded in field experiments (Mitsch et al. 1999), but for the 10 million acres of restored wetlands to average this level of N retention would not be expected. However, a relatively small increase in filtering efficiency for the mixed policy would further enhance its attractiveness.

An important point of caution in interpreting the results is that the information on environmental benefits from restoring wetlands and reducing nitrogen and sediment runoff is limited. The values we used most likely underrepresent the use and nonuse values associated with the nitrogen-reduction strategies we present. If all the benefits could be accounted for, the final assessment might be different.

Finally, we want to stress the importance of institutional factors, administration, monitoring, verification, and enforcement (whether by regulation or by markets). These all become even more critical as multiple strategies are employed requiring different combinations and levels of each of the above to successfully implement a strategy.

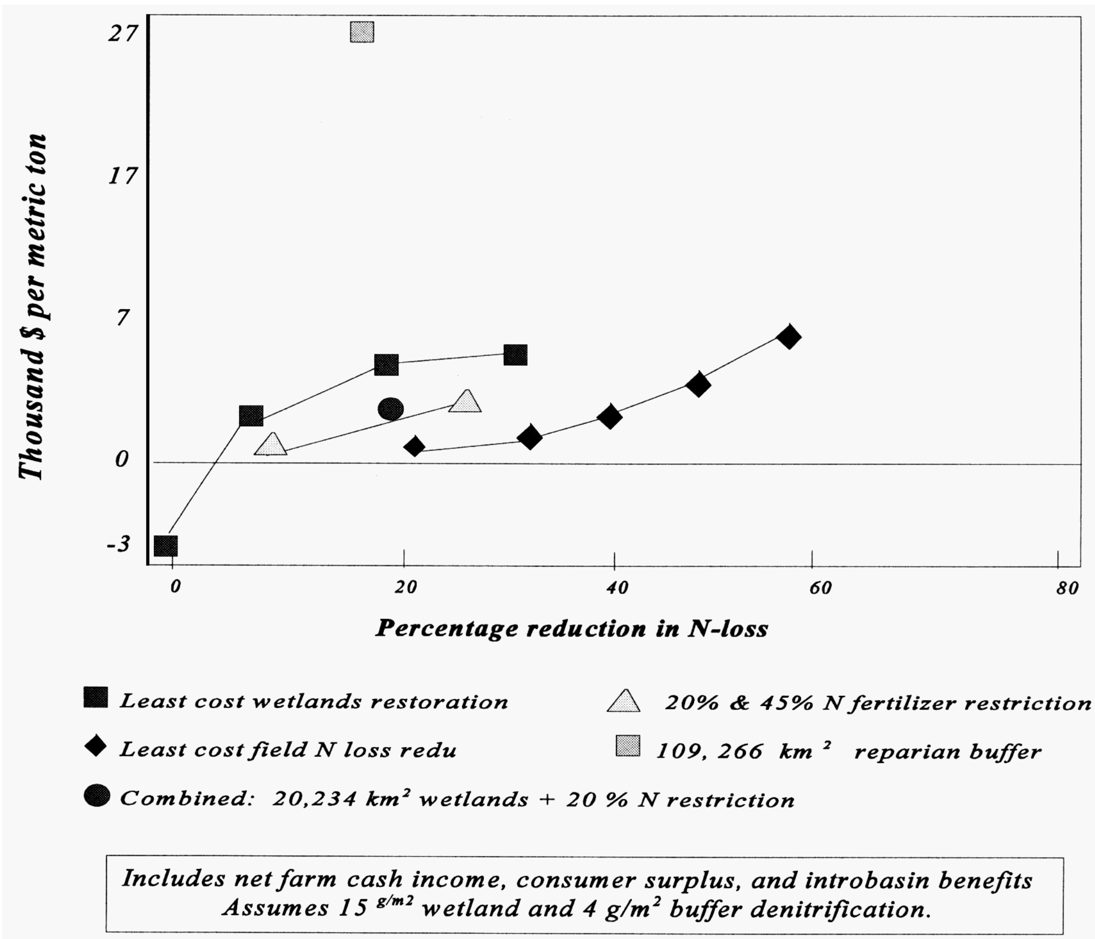

FIGURE 6.1 Agricultural sector adjustment costs per unit of N reduction, net of intrabasin benefits, by control method.

CHAPTER 7

Institutional Considerations

7.1 TRANSACTION COSTS

Any policy mix to reduce agricultural nonpoint pollution in the Mississippi River system feeding the Gulf of Mexico implies a "cost of doing business"—a transaction cost. In fact, all market transactions entail a cost of gathering information, negotiating, and staking claim to the result (Schmid 1995). Randall (1975) defines transactions costs as "costs of resolving situations where involved parties have conflicting interests . . . including the cost of gathering information, determining their position and strategy; the cost of bargaining, negotiating, arbitration, judicial or other process by which agreement is reached . . . ." Distribution of that cost is a reflection of the relative power of market participants, defined in property rights and other market rules. If government, acting on behalf of some water users who are able to attain the status of "public interest," alters the economic signals facing other water users through a regulatory or incentive mechanism to bring about changes in decisions to affect water quality, there are costs of enforcement, monitoring, and other steps required to ensure that the policy change has the desired effect. The total cost and its distribution among affected parties will vary with the technique selected. The management boundary chosen also affects that cost by influencing how those who gain from pollution and those who gain from abatement reconcile their differences.

7.1.1 Hydrologic versus Political Boundaries

Hydrologic boundaries have long been accepted as the appropriate unit for managing complex river systems. Comprehensive river basin planning undertaken for the large river systems by the U.S. Army Corps of Engineers in the 1960s and by USDA for smaller upstream river segments emphasized watersheds as the management unit. President Kennedy drafted legislation in 1961 to establish a Water Resources Council of federal agencies having water-related responsibilities and to establish river basin planning commissions to undertake comprehensive basin planning. These proposals followed recommendations of an earlier Senate Select Committee on National Water Resources (Holmes 1979). Although final approval of the basin planning process did not occur until 1965, interim procedures moved ahead and 20 framework and basin plans were under way when the Water Resources Planning Act of 1965 was finally enacted (Holmes 1979). President Lyndon Johnson submitted the first national water assessment to Congress in 1968, based on river basins and sub-basins (U.S. Water Resources Council 1968). Local watershed protection efforts were supported through the Soil Conservation Service beginning in the 1950s (Holmes 1979). The Clean Water Action Plan reinforces that priority in current policy.

Hydrologic boundaries acknowledge the reality that water flows downhill, and in so doing binds all people and activities along the way. The question is whether human communities can organize their affairs to respond to the physical reality of a watershed. So far, we have not been particularly successful. We have organized ourselves differently for convenience of governance, service delivery, and general identity or location. People live and pay taxes in municipalities, towns, counties, and states—not watersheds. Water quality is only one set of services that people care about. The practical problem of managing river systems requires the costly and often frustrating process of cutting across or further dividing political jurisdictions.

The City of Gainesville, Florida, for example, is divided by two very large river basins, each of which is served by a Water Management District with taxing authority. The St. Johns is to the east, the Suwannee to the west. There is no obvious line separating the basins, and residents expecting basically the same services from their local governments pay different taxes. Presumably the water quality services are different between the two basins, but those differences are not apparent to residents.

The Mississippi system, the nation's largest, encompasses all or part of literally thousands of political jurisdictions that are connected by physical reality, though little else. There are many sub-basins in the Mississippi that are less complex with more obvious internal linkages, but the basic incongruence between hydrologic and political boundaries remains.

The Clean Water Action Plan places major emphasis on states (political jurisdictions) to develop nutrient standards and criteria that acknowledge important differences among aquatic eco-systems (hydrologic). There will be national guidelines and oversight, but state action. The cost of improving water quality will be shared, but the federal government will provide the major portion through Section 319 grants and with technical assistance to recognize that benefits of clean water are broadly distributed. Under the Clean Water Act, EPA is authorized to spend $130 million a year in cost sharing for nonpoint abatement (Sohngen and Taylor 1998).

Success for the Gulf of Mexico and the Mississippi system depends on the degree of consistency from state to state in parts of each sub-basin and basin. It will require collaboration and negotiation between upstream and downstream jurisdictions within states and among states from one end of the basin to the other. Each political jurisdiction will perceive a different stake in the water-use decisions made, depending on their location and a different trade-off between water quality and other goals. These transaction costs are an inevitable part of water quality management, formalizing the adage that we think globally (Gulf of Mexico) and act locally. Costs are particularly high with nonpoint abatement, where polluters and their relative contributions to the problem are poorly defined.

7.1.2 Regulations and Incentives

Water quality improvement since the passing of the 1972 Clean Water Act has relied primarily on required abatement technologies. Once regulators know that "best available technologies" or best management practices are in place, they are confident that the best possible effort to control pollution is being made. But these best practices often overlook opportunities to replace whole production systems at a lower cost and thus may stifle innovation (Tietenburg 1985). Implementation requires checking the water users to be sure they are installing required technologies, penalizing those who do not, and then dealing with any appeals. In the case of nonpoint pollution, there is stiff resistance to the basic idea of federally mandated farming practices and the practical problem of checking compliance by thousands of farmers and loggers. Both types of implementation are expensive.

Alternatively, performance-based regulations do not mandate specific practices and instead require achievement of defined water quality standards by whatever means the water user chooses. This approach requires extensive monitoring of water quality by river segment or by individual water user, holding all users responsible for the result. Innovative trading schemes are being considered to reconcile differences in compliance costs among types of water users. A polluter for whom compliance with a standard would be very expensive may buy a right to pollute from a user who can reduce pollution much more cheaply. Thus, overall water quality is protected through efficient trades among polluters, some of them point and others nonpoint sources (Apogee Research 1992). The transaction costs of these trading systems can be very high, in setting up the market for pollution rights and designing appropriate trading ratios that control for the uncertainty inherent in nonpoint abatement techniques. There is the danger of allowing the trade and finding that water quality has not improved at all. A trading ratio that is too high (requiring too much of the purchaser) may discourage trades, thus having no effect on water quality (Sohngen and Taylor 1998).

Pollution fees involve levying a tax on a certain increment of pollution, creating an incentive to reduce production costs by reducing pollution. Transaction costs in all incentive programs relate primarily to the cost of information. Success in reducing pollution requires that the tax be high enough to induce behavior change but not so high as to be unnecessarily punitive. If the fee is too low, the rational water user will just pay the fee and continue polluting. Data must be collected on the water quality impacts of a given tax structure.

CHAPTER 8

Concluding Observations

8.1 BENEFITS FROM REDUCING NITROGEN LOSS IN THE GULF AND MISSISSIPPI BASIN

The direct benefits to Gulf fisheries of reducing nitrogen loads from the Mississippi River basin are very limited at best. (See the Topic 2 economic analysis.) We do not have the information necessary for estimating the benefits of such factors as restoring the ecological communities in the Gulf, which over the long term may be significant.

In most cases, indirect cost savings to agricultural producers from reductions in fertilizer nutrients to cropland were modest. For the agricultural sector, commodity prices and aggregate producer net returns increase at increasing levels of reduction in nitrogen loss. This is not a benefit from nitrogen reduction directly, but derives from reduced production resulting from reduced fertilizer application. These begin to be significant when nitrogen-loss restrictions reach 30% and higher. One difficulty with these benefits is the problem of accounting stance (the question of whose benefits and costs we are enumerating). Severe restrictions on nitrogen loss from agriculture mean that production ceases on acres especially vulnerable to nitrogen loss and causes shifts to cropping systems that lose less nitrogen. Production of crops with high nitrogen losses is also increasingly shifted out of the basin. Some producers suffer these losses, while those remaining in production with cropping systems that provide relatively high value benefit from increased commodity prices as the supply is reduced due to nitrogen restrictions. Aggregate returns to agriculture increase, but costs are imposed on some who are constrained to abandon profitable production in order to meet nitrogen-loss goals.

The values of potential benefits from nitrogen reduction within the basin are potentially significant. Even a modest program of fertilizer reduction and wetland restoration is estimated to produce benefits within the basin. These include the values associated with restored wetlands; improved recreational water quality; and reduced soil erosion, nutrient contamination of drinking water, and water-borne human pathogens from changes in cropping systems and livestock production.

8.2 COSTS OF REDUCING NITROGEN LOSS IN THE GULF AND MISSISSIPPI BASIN

The Topic 2 economic evaluation team concluded that reducing nitrogen loss may cause shifts in species composition, which may result in costs to the recreational and commercial fishing industries. However, the available information is inadequate for estimating these costs.

As nitrogen is progressively limited, or as acres are taken out of production for wetland restoration, agricultural production within the basin and nationally is reduced and crop prices increase, affecting consumers. Changes in production practices to meet program goals also impose some adjustment costs on the agriculture sector. Table 4.7 presents the economic impacts on producers and consumers for different levels of nitrogen reduction and for different levels of wetland restoration and buffers.

We found only modest aggregate impacts on the agricultural sector for up to a 20% reduction in nitrogen loss (comparable to the 15–20% reduction in nitrogen losses from agriculture deemed feasible and recommended by the Topic 5 team). We found that restoring 5 million acres of former wetlands also had a minimal impact on agricultural production and related factors. At the 10-million-acre level, price, land-use, and other impacts began to be noticeable. The Topic 5 team was primarily concerned about producers' ability to achieve nitrogen reductions using feasible production practices. Our analysis accounted for the economic impacts on the producers and kept changes in acreage and exports well within the historic bounds of recent adjustments—a matter of concern to many in the agricultural sector.

Livestock producers would bear higher feed grain costs as prices increase under nitrogen-loss restrictions. Consumers of basic commodities and the finished food and fiber products derived from them would suffer some loss from price increases caused by production changes and acreage restored to wetlands.

One factor of concern that we were able to measure was increased nitrogen, phosphorus, and sediment loss in agricultural production that occurs outside the basin as nitrogen use is restricted within the basin. Reducing nitrogen losses to the Gulf is likely to impose additional pollution costs on the rest of the nation as an indirect impact. Price increases due to reduced production within the basin result in an intensification of crop production elsewhere. These potential environmental impacts are presented in Figures 4.3–4.5 for different levels of nitrogen reduction. Again, the extent of derived environmental impacts was very modest up to the 20% nitrogen-loss reduction level.

There is also a potential cost from decreased agricultural export volumes that depends upon the level of nitrogen restriction (although the value of exports would increase because of price increases). Changes in export volumes were estimated for different levels of nitrogen-loss reduction. Again, these reductions become more important and begin to break out of historical bounds at and above the 30% nitrogen-loss restriction level. At 30%, corn exports are down 11% and wheat exports are down 23%; at 60%, corn and wheat exports are down 35% and 65%, respectively. The primary concern of the agribusiness industry is loss of sales in an expanding free-market environment, where market share is voluntarily constrained to meet environmental objectives. Also, reduced acreage in production and reduced output can have negative impacts on input and shipping sectors.

Social costs would also be incurred, such as dislocation in land use, agribusiness infrastructure, and farm communities. We can tell in some cases, and infer in others, where we might begin to incur unacceptable costs of this kind on the basis of previous historical shifts in crop production, land use, and input use. We did not estimate these costs.

The case studies give some indication of the costs agricultural producers would face in reducing nitrogen losses. The aggregate analysis was driven by the criterion of profit maximization, so the strategies chosen by producers in the model were those that either improved or did the least damage to their incomes. However, specific producers might incur severe costs, while others might benefit from reduced costs if constrained to lower nitrogen losses.

For atmospheric deposition, we have presented previous case studies of the costs of adopting nitrogen-reducing management or technology. Reducing nitrogen emissions from mobile sources appears to be more costly than reducing emissions from utilities, and both of these strategies appear to be more costly than initial efforts focusing on nonpoint sources when they are at the stage of eliminating easily reduced nitrogen losses.

For the reduction of nitrogen from traditional point sources in the basin, we have analyzed the costs of nitrogen restrictions on the basis of existing plant-level technology, compared with the option of trading between point and nonpoint sources to achieve reductions. The least-cost approach in roughly half of the cases is for point sources to purchase nitrogen reductions from nonpoint sources.

Administration, monitoring, verification, and enforcement costs are very real and important, but have not been included here. They must be included as specific policy approaches are chosen for analysis. For example, one of the keys to nonpoint-source reductions will be adequate verification that the reduction is in fact taking place. This is especially critical if point sources trade for reductions with nonpoint sources. For nonpoint-source reductions from agriculture, this will be confounded both by the time lag inherent in soil systems' behavior and by the current lack of monitoring capacity.

8.3 COST-EFFECTIVE STRATEGIES

The bottom line is that reducing nitrogen losses in the 20% range is feasible, and there are relatively cost-effective ways to achieve this goal.

- Wetland-based strategies are more expensive than fertilizer-reduction strategies to achieve the same nitrogen-loss reduction goal. Land-retirement costs and wetland-restoration costs outweigh the higher environmental benefits generated by wetlands.
- Vegetative buffers are not very cost-effective for the specific task of reducing the nitrogen losses we are concerned with here, due to low nitrogen filtering relative to wetlands, lower wildlife-associated benefits, and high land-retirement costs.
- Fertilizer restrictions are more cost-effective than a fertilizer tax, due to the tax's impacts on producers' net returns.
- A strategy that combined a 5-million-acre wetland restoration goal with a 20% fertilizer-reduction goal was the most cost-effective, practicable approach for meeting a 20% nitrogen loss-reduction goal. Reducing fertilizer application by 45% met the 20% goal at a slightly higher cost. A policy that includes wetlands has additional advantages because it meets other policy objectives and probably generates wildlife and recreation benefits greater than those estimated here. Figure 8.1 summarizes this strategy, comparing its costs and benefits.

The results of this report are based on estimates of wetland- and buffer-filtering capacities, and estimates of environmental benefits that are crude at best. The research in these areas is sparse and incomplete.

Finally, cost-effectiveness depends upon the actual delivery of nitrogen at the point of concern. When the objective is reducing nutrients delivered to the Gulf, the critical issue is going to be the relationship between an action upstream and what actually comes out of the mouth of the river. Because such nonpoint sources as agriculture and the interacting soil systems represent a large volume of the absolute nitrogen in the system, a given percentage reduction of nitrogen loss from agriculture within the basin may result in a different proportional decrease in the amount of nitrogen flowing into the Gulf.

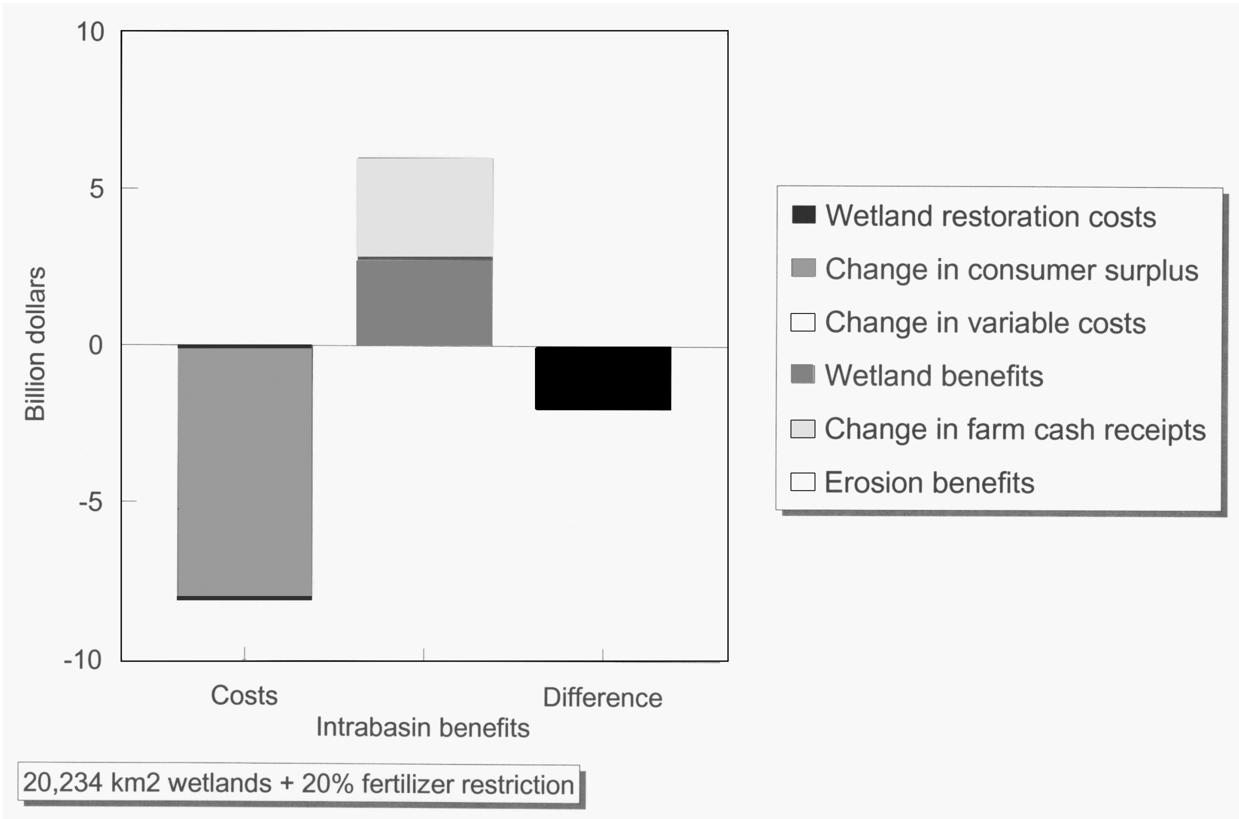

FIGURE 8.1. Illustrative costs and benefits of nonpoint nitrogen control in the U.S.

APPENDIX A

Animal Waste

Livestock manure is believed to be a significant source of nutrient-related water quality problems. Manure is enriched primarily with nitrogen, phosphorus, and potassium. If these nutrients are oversupplied to fresh and coastal waters, eutrophication and hypoxia can occur. According to the U.S. Geological Survey (USGS), livestock manure contributed approximately 20% of the total nitrogen and 50% of the total phosphorus loads in 1997 within the Mississippi Basin. The Topic 3 report examines the production of "recoverable" livestock manure within the Mississippi watershed and its effects on water quality (Goolsby et al. 1999).

The increasing concentration of livestock within the Mississippi Basin has degraded its water quality. Although leaks from the pits and lagoons used to store livestock waste also contribute to water quality problems, they are not the focus of this analysis. It is assumed that all manure storage units were constructed efficiently and that water degradation is only a result of inefficient land application of livestock waste and the problems associated with the nutrients supplied by this waste.

A.1 OVERVIEW OF LIVESTOCK PRODUCTION

Over the past 40 years, manure has continued to be a relatively constant nitrogen input to the Mississippi–Atchafalaya Basin (Figure 5.7 of Goolsby et al. 1999). During this same period, the structure of the various livestock industries has changed, resulting in fewer but larger farms. Although the number of farms has decreased, overall production has remained stable. The livestock industry has become more specialized and intensive with the emergence of confined feeding operations, thus creating greater volumes of liquid and solid manure in smaller areas (Sutton and Jones 1998).

A detailed analysis of the changes in livestock production by McBride (1997) showed that in most animal sectors the number of animals per farm substantially increased between 1969 and 1992 (Table A.1). For further discussion of the increased concentration of the livestock industry, *Food System 21* (1997) continues with data up to 1996.

The total number of animals in the basin has not changed significantly over time, thus explaining the constant input of nitrogen over the past 40 years. The main issue related to water quality degradation is associated with the increase in concentration of animals per area of production. In regions of high concentration, the amount of nitrogen produced is often greater than what the crops are able to absorb (Lander et al. 1998). Nitrogen that is not taken up by crops leaches into ground water, is captured in tiles, or is carried in runoff to surface water. This excess nitrogen degrades water quality.

TABLE A.1. Number of head per farm: 1969 and 1992.

Animal Type	1969	1992	% Change
Hogs	138	588	326
Milk Cows	20	61	205
Fed Cattle[1]	106	154	45
Beef Cow	41	40	-2
Broiler	70,798	237,622	236
Layer	632	2985	372

[1]Data for fed cattle are from 1978–92.
Source: McBride 1997.

The Mississippi Basin has approximately 31,893,250 animal units[4]. (For more specific analysis, Figure 5.7 of Goolsby et al. (1999) breaks down the basin into six sub-basins: the Ohio, Tennessee, Upper and Lower Mississippi, Missouri, and Arkansas–White–Red.) Table A.2 shows that the Upper Mississippi and Missouri have the greatest number of animal units among the sub-basins within the Mississippi watershed, and that the Arkansas–White–Red has the highest concentration of animal units per farm.

TABLE A.2. Number of farms and animal units by sub-basin in the Mississippi watershed.

Sub-basin of the Mississippi Watershed	Number of Farms	Total Animal Units	Estimated Animal Units per Farm
Ohio	73,901	5,187,950	70.2
Tennessee	14,289	1,255,000	87.8
Upper Mississippi	92,259	10,503,700	113.9
Lower Mississippi	12,377	1,203,800	97.3
Missouri	76,441	8,388,700	109.7
Arkansas–White–Red	37,007	5,354,100	144.7

Source: U.S. Mathematical Programming Model analysis, Economic Research Service, USDA.

[4]Measuring the number of animals by animal units allows for more accurate analysis. It assists in determining how much manure is produced within the Mississippi watershed by all animals. An animal unit (AU) is defined as 1,000 pounds of liveweight of any given livestock species or any combination of livestock species. AU equivalents are calculated for each livestock and poultry sector according to estimated rates of manure production per species. Thus, AUs vary by sector: 1 beef cow is equivalent to 1 AU, while 250 layers equal 1 AU.

A.2 EFFECTS OF LIVESTOCK PRODUCTION ON WATER QUALITY

Water quality problems associated with animal manure occur when more nutrients are supplied to the land than what the crops can use for growth. For the Mississippi Basin the estimated area of land suitable for application of manure is, 82,487,500 acres. Table A.3 estimates the acreage available for manure application within each sub-basin. Tables A.4 and A.5, respectively, estimate the volumes of nitrogen and phosphorus supplied within each sub-basin from the application of manure to cropland.

TABLE A.3. Estimated area of land for manure application.

Miss. Sub-basin	Acres of Land
Ohio	15,033,300
Tennessee	1,676,700
Upper Mississippi	25,602,850
Lower Mississippi	1,895,750
Missouri	29,642,250
Arkansas–White–Red	8,636,650

Source: U.S. Mathematical Programming Model analysis, Economic Research Service, USDA.

TABLE A.4. Estimated manure nitrogen production.

Miss. Sub-basin	Pounds
Ohio	336,492,200
Tennessee	86,888,150
Upper Mississippi	661,935,850
Lower Mississippi	77,262,200
Missouri	514,767,700
Arkansas–White–Red	339,708,850

Source: U.S. Mathematical Programming Model analysis, Economic Research Service, USDA.

TABLE A.5. Estimated manure phosphorus production.

Miss. Sub-basin	Pounds
Ohio	119,807,150
Tennessee	29,250,700
Upper Mississippi	233,607,850
Lower Mississippi	28,119,600
Missouri	185,531,900
Arkansas–White–Red	113,991,700

Source: U.S. Mathematical Programming Model analysis, Economic Research Service, USDA.

A.2.1 Studies of Nutrient Overload

Two nationwide studies were conducted focusing on the supply of nutrients from manure and land available for spreading. The resulting analyses of these studies identified areas where more nutrients were available than what the land could absorb. The first study, by Letson and Gollehon (1996), looked at the areas within the U.S. with high nutrient application. They concluded that the Pacific, Southeast, Delta, and Appalachian regions have high total amounts of nutrients. Therefore, nutrient management policies could focus on these areas in order to potentially reduce water quality impacts from overabundant nutrient application. This study found that often when the amount of cropland per farm did not match the need for land application of animal manure, it would tend to lead to a higher probability of water quality degradation.

The second study, by Lander et al. (1998), estimated the nutrient availability from manure and crop nutrient uptake and removal for each county in the 48 states. From these estimates, they calculated nitrogen and phosphorus availability relative to crop need. They concluded that areas exist where more manure is produced relative to the land available for application. Therefore, these results could assist in identifying areas likely to have water quality problems.

Letson and Gollehon (1996) estimated the pounds of recoverable and total nutrients that are applied per acre of land suitable for spreading animal manure. This information, applied to each sub-basin, is illustrated in Table A.6. The total amount of nutrients was calculated from the total manure produced. Recoverable nutrients were calculated based on the economically recoverable manure from animals. The total amount of manure excreted cannot be entirely collected for land application. Therefore, the amount that can be economically gathered for disposal is referred to as “recoverable” manure. It is from the recoverable manure that the recoverable amount of nutrients is calculated.

On average, the recoverable amount of N that is spread per acre within the entire Mississippi watershed is 15.69 pounds. The average total amount of N that is spread per acre within the basin is 25.07 pounds. The important result to derive from these numbers is they enable us to determine the quantity of nutrients applied at a rate that cannot be fully absorbed. To determine the impact on water quality it is necessary to identify what percent of these manure applications exceeds the amount of nutrients needed.

TABLE A.6. Nutrients applied per acre from manure.

Miss. Sub-basin	Recoverable P Spreadland	Recoverable N Spreadland	Total P Spreadland	Total N Spreadland
	Pounds per Acre			
Ohio	5.79	12.55	7.59	20.98
Tennessee	13.49	29.17	15.79	46.75
Upper Mississippi	6.99	14.53	9.61	26.04
Lower Mississippi	9.59	19.75	11.98	34.32
Missouri	4.19	9.04	5.06	14.25
Arkansas–White–Red	12.71	28.48	14.22	42.00

Source: Letson and Gollehon 1996.

Lander et al. (1997) estimated the percent of manure applications that exceed the amount of nutrients needed for each sub-basin (Table A.7). For instance in the Tennessee sub-basin, almost 43% of the total manure land applications exceed the amount of N needed for that basin, and over 66% of the applications in the same sub-basin exceed the amount of P needed. On average, the entire Mississippi Basin exceeds the amount of N needed 20% of the time and exceeds the amount of P needed 43% of the time.

TABLE A.7. Potential for nitrogen available from animal manure to meet or exceed plant uptake and removal.

Miss. Sub-basin	N Balance (%)	P Balance (%)
Ohio	13.59	32.03
Tennessee	42.97	66.03
Upper Mississippi	11.77	41.13
Lower Mississippi	41.21	41.18
Missouri	7.21	27.50
Arkansas–White–Red	44.06	79.13
Mississippi Basin (avg.)	20.00	43.00

Source: Lander et al. 1997.

According to the data, it is evident that within the Mississippi Basin more nutrients from manure are being applied to the land than what can be absorbed. Nitrogen and phosphorus are being overapplied, thus contributing to water quality problems. Phosphorus tends to be more of a problem according to the data. Therefore, many of the issues that arise from manure disposal focus on phosphorus-related issues.

A.2.2 Water Quality and Manure

Negative impacts from animal agriculture on the environment have stimulated public concern in recent years. Nutrients from manure stimulate excessive growth of aquatic vegetation, which leads to eutrophication and hypoxia. In addition, soluble nitrogen has leached into aquifers, thereby damaging local drinking-water supplies. Pathogens and bacteria, also in manure, can impair surface waters used for drinking and recreation (Letson and Gollehon 1996).

Many of the public and political concerns that focus on animal manure originate from fresh-water impairments. Because phosphorus is the limiting factor in freshwater cultures, it is an important component for stimulating vegetative growth in water bodies. When P is oversupplied, this vegetation grows at an excessive rate, which depletes dissolved oxygen and reduces biodiversity. It is common for P to be oversupplied when manure is applied based on the N needs of a crop. “Long-term application of manure, even at rates not exceeding the nitrogen needs of crops, can still result in soil buildup of P and K” (Sutton 1998). This buildup of P can leach into ground water or can be transported in runoff and result in water quality degradation. States are beginning to consider changing land-application requirements to be based on the crop’s P needs instead of its N needs. If this occurs, more acreage will be needed for land application, and the overapplication of N will be reduced (Sutton 1998).

Since 1990, manure management has also been a concern for national policymakers. The 1990 Coastal Zone Act Reauthorization Amendments (CZARA) was the first legal effort to address nonpoint-source pollution from agriculture by focusing on manure management. In addition, the Environmental Quality Improvement Program (EQIP) also encourages improved management of animal waste. Most recently a joint effort of the U.S. Department of Agriculture (USDA) and the U.S. Environmental Protection Agency is focusing on the Unified National Strategy for Animal Feeding Operations (AFOs), where manure management plans will be formulated for all AFOs in order to reduce water quality impacts from animal agriculture. Other regulations being considered are requiring that manure be applied to land based on the crops' phosphorus needs, as opposed to their nitrogen needs.

Pathogens and coliform bacteria originating from animal manure are important factors for water-related health problems, although for the issue of hypoxia in the Gulf of Mexico we are concerned only with the nutrients supplied by manure.

A.2.3 Manure Management

Manure contains the nutrients nitrogen (N), phosphorus (P), and potassium (K), which enhance both crop production and the chemical and physical properties of soil. The type and amount of nutrient contained in livestock manure vary, depending on animal type, amount and type of feed, method of storage, time and method of land application, soil characteristics, crops present, and climate.

Each type of animal has a different nutrient content in its manure. Because poultry manure has the highest nutrient content, it has been found to be a suitable input for cattle feed (Mitchell and Donald 1995; Ruffin and McCaskey 1997). The numbers listed in Table A.8 vary between farms; thus, they are based on the average nutrient excretion from animals. For instance, the amount and type of feed a farmer uses can greatly influence the amount of nutrients excreted.

TABLE A.8. Annual pounds of nutrients in manure, as excreted per 1,000 pounds liveweight.

Nutrient	Dairy Cow	Beef Feeder	Swine Feeder	Laying Hen	Broiler
Nitrogen, N	150	124	164	263	423
Phosphate, P_2O_5	60	91	124	232	216
Potash, K_2O	118	106	132	136	136

Source: Hammond et al. 1994.

The method of storage also affects the N concentration of manure (Table A.9). N loss mainly occurs when it has been exposed to the sun and or air, which would occur in an open-lot system. When manure is stored in pits beneath the animal housing there is less N loss, although N loss is greatest in treatment lagoons.

TABLE A.9. Nutrient losses from animal manure, as affected by method of handling and storage.[1]

Manure Handling and Storage Method	Percent N Loss	Percent P Loss	Percent K Loss
Solid Systems			
Daily Scrape and Haul	15–35	10–20	20–30
Manure Pack	20–40	5–10	5–10
Paved Lot	40–60	20–40	30–50
Deep Pit (poultry)	25–50	5–15	5–15
Litter (poultry)	25–50	5–15	5–15
Liquid Systems			
Anaerobic Deep Pit	15–30	5–15	5–15
Above-ground Storage	5–25	5–15	5–15
Earthen Storage Pit	20–40	10–20	10–20
Lagoon	70–85	50–85	50–75

[1]Based on composition of manure applied to the land vs. composition of freshly excreted manure, adjusted for dilution effects of the various systems.
Source: Sutton et al. 1994.

The method used for applying the manure on the land will also cause nitrogen loss (Table A.10). It is evident from this information that when the manure is incorporated, N loss to the atmosphere is significantly reduced. Incorporation also allows soil microorganisms to start decomposing the organic matter in the manure, thus making nutrients available to the plant sooner. P and K have been found not to be significantly affected by the application method; however, incorporating manure minimizes P and K losses from surface runoff. If manure is applied before crops are planted, nutrient availability is maximized (Hammond et al. 1994).

TABLE A.10. Approximate nitrogen losses from manure to the air, as affected by application method.

Application Method	Consistency of Manure	Nitrogen Loss[1] (%)
Broadcast without Incorporation	Solid Liquid	15-30 10-25
Broadcast with Incorporation[2]	Solid Liquid	1-5 1-5
Knifing (injection)	Liquid	0-3
Irrigation	Liquid	30-40

[1]Percent of total nitrogen in manure applied that was lost within four days after application.
[2]Cultivation immediately after application.
Sources: Hammond et al. 1994; Sutton et al.

A.2.3.1 LAND APPLICATION EXAMPLE

When applying manure to the land it is important to know the rate at which the waste should be spread. To reduce water quality impacts, the rate of application should be based on the crop's nutrient needs. Current state regulations require that manure be spread at a rate based on the nitrogen requirements of crops (Sutton and Jones 1996). The application rate to spread the manure depends on the yield of the crop, the nutrient content of the manure according to the method of storage, and how the manure will be spread. All of these factors will affect the amounts of nitrogen available and needed for the crop. Through the use of AMANURE, a computer simulation program produced by Purdue University, the amount of acreage needed can be determined for a typical hog farm. In Table A.11 the acreage needed per sow was determined for corn (130 bu/ac) soybean rotation, a 15-foot swath, and incorporation. The acreage needed under the lagoon system is significantly less than for storage pits because of the high rate of nitrogen lost from storing manure in lagoons (Table A.9).

TABLE A.11. Acreage and application rate for manure per sow.

Storage Method	Acres/Year Needed for Land Application Based on:		Application Rates, Gallons/Acre Based on:	
	Nitrogen	Phosphorus	Nitrogen	Phosphorus
Pits	0.6	1.1	7,431.4	4,294.9
Lagoon	0.1	0.2	57,186.3	34,174.8

The nutrient content of manure is not balanced. Animal manure contains a higher amount of N than of P (Table A.8), but a crop such as corn requires proportionally more nitrogen than phosphorus. Therefore, when manure is spread at a rate based on N, P tends to be applied above crop needs (Fleming et al. 1998). To combat this overapplication of P, policymakers may move toward requiring application of manure to be based on P rather than on N. A few studies have estimated the impact of this potential policy change. Fleming et al. (1998) concurred that "basing manure applications on phosphate levels rather than nitrogen increases the value of manure nutrients because applied nutrients better match crop requirements. This also increases the cost of delivering hog manure because application rates on corn are reduced; hence more acres must be used for application." From the AMANURE results the increase in acreage requirements per sow almost doubles (Table A.11), illustrating Fleming's conclusion.

With the increased acreage requirement for land application based on the phosphorus needs of the crop, it is useful to discuss the potential changes that could occur on a farm. With the use of AMANURE, Table A.12 illustrates the number of acres needed for land application based on the herd size of a hog operation. The simulation again assumes a corn (130 bu/ac) soybean rotation, a 15-foot swath, and incorporation. Fleming et al. (1998) also concluded that "as herd size grows, however, costs of following a P-standard rise faster than the costs of following an N-standard; thus, following an N-standard on continuous corn would eventually minimize net losses. Generally, marginal delivery cost increases because more acreage is needed to apply the same quantity of manure (mileage increases)." With continued concentration and growth of the livestock industry, manure management issues need to be considered.

TABLE A.12. Total acres needed per year for land application of manure based on method of storage and nutrient constraint.

Herd Size	Storage Pits		Lagoon	
	Nitrogen	Phosphorus	Nitrogen	Phosphorus
100	61.1	105.8	13.5	22.6
200	122.3	211.6	27.1	45.3
300	183.4	317.4	40.6	67.9
500	305.7	528.9	67.6	113.2
750	458.5	793.4	101.5	169.8
1,000	611.4	1,057.3	135.3	226.4

Expansion issues arise when assessing whether enough land will be available for economical spreading. Does the farmer have more land available on site that could support the extra nutrients available? Can he apply them to the neighbor's field? Can he rent or buy more land? Can he give the extra nutrients to a custom hauler for ecologically safe disposal? The important consideration, if expansion in the livestock industry occurs, is land available for effectively disposing of the extra nutrients to avoid further water quality problems. If no economical land is available, alternative methods for disposing of manure may need to be explored.

Waste management systems for dairy and hog operations typically use lagoon and liquid tank (pit) methods for storage. A study conducted by Bennett et al. (1994) reported the annual costs of operating lagoon and pit systems, which included costs for spreading manure. The study focused on dairy waste management systems in Missouri, and the data are used here to provide an example of representative costs a farmer would incur for manure disposal. There are also benefits to waste management, which are determined by the market value of the manure's nutrients.

Lagoons are more commonly used in large hog and dairy operations because labor and investment costs are minimized and a flushing system can be used to collect and transport waste to the lagoon (Bennett et al. 1994). Manure is disposed of through an irrigation system pumped directly from the lagoon through a pipe, and spread on the land through irrigation equipment. Bennett's study uses a traveling gun, but other irrigation methods are available for farmers. In Table A.13 annual operating costs are listed for a dairy lagoon system in Missouri. The data assume that the farmer hires a custom irrigation system but owns other equipment. Bennett et al. found that when comparing the use of owned versus custom-hired traveling gun irrigation systems, it becomes economical to use custom-hired systems after the herd size exceeds 300 head.

Liquid tank or pit systems are often used in smaller or medium-sized hog and dairy operations. Because manure is less liquid in this form, it has a higher market value due to a higher concentration of nutrients. Manure can be scraped for collection, or it is often required to agitate the manure in order to pump it into a tank wagon for spreading. Table A.14 lists the costs required for waste management, which includes the scraping, agitating, pumping, and spreading costs. The spreading cost is included in the use of the tank wagon, which disposes of the waste on the land. Bennett et al. concluded that the lagoon system becomes economical once the herd size exceeds 500 head.

TABLE A.13. Annual operating and disposal costs of Missouri dairy lagoon system.

Cost Factors	Herd Size					
	100	200	300	500	750	1,000
1. Size of electric pump (hp)	1.5	1.5	3.0	5.0	7.5	7.5
2. Annual pumping time (5 hrs. daily x 365 days)	1,825	1,825	1,825	1,825	1,825	1,825
3. Cost/hour (1 kw x hp x 0.7/kw)	$0.11	$0.11	$0.21	$0.35	$0.53	$0.53
4. Irrigation lagoon annual pumping cost (line 2 x line 3)	$201	$201	$383	$639	$967	$967
5. Estimated acre-inches to pump annually	57	85	106	155	204	260
6. Custom pumping charge/hour[1]	$60	$60	$60	$60	$60	$60
7. Annual lagoon pumping cost (line 5 x line 6)	$3,420	$5,367	$6,821	$10,053	$13,363	$16,765
8. Check labor hours/year[2]	7	11	13	19	26	33
9. Annual labor costs (line 8 x $6/hour)	$42	$66	$78	$114	$156	$198
10. Total operating costs (line 4 + line 7 + line 9)	$3,663	$5,367	$6,821	$10,053	$13,363	$16,765

[1]Assume a traveling gun pumps 500 gallons per minute, which allows pumping one acre-inch per hour.
[2]Check labor required to inspect the irrigation system periodically to determine if the traveling gun and equipment are operating adequately. Assume one hour per eight hours of operation.
Source: Bennett et al. 1994.

The market value or benefit received from delivered nutrients will depend on both the amount of nutrients used by the crop and the price of nutrients in the location of the farm. Table A.15 lists the prices received for nutrients in Iowa, Indiana, and Missouri. The true cost of manure disposal should figure in the benefit a farmer receives in fertilizer value from the manure.

Manure disposal cost depends on the storage and application methods and on the nutrient content of the waste. The cost to farmers will vary but the above example can provide an idea of the types of costs and benefits farmers incur when faced with manure disposal issues. Other issues that need to be considered are the timing of the manure application and the trade-off of how farmers will use the available labor time. A farmer may have more incentive to hire custom spreading services during planting time due to the high opportunity cost of labor. Land available for spreading is also an issue to consider. To avoid overapplication, enough land needs to be available to recycle the nutrients supplied by the manure. This issue is key to avoiding water quality damages originating from manure disposal practices.

TABLE A.14. Annual operating costs of Missouri dairy liquid manure tank (pit) system.

Cost Factors	Herd Size					
	100	200	300	500	750	1,000
Power Supply, Hours Used Annually						
1. Scraper tractor, 40 hp	183	304	426	669	852	1,278
2. Agitation pump tractor, 100 hp[1]	7.2	10.8	14.5	21.9	31.0	40.2
3. Tank wagon tractor(s), 100 hp[2]	172	260	406	699	1,118	1,610
Power Costs						
4. Scraper tractor (line 1 x $7.62/hr)	$1,394	$2,316	$3,246	$5,098	$6,492	$9,738
5. Agitation and wagon tractor (line 2 + line 3 x $19.36/hr)	$3,469	$5,242	$8,140	$13,957	$22,245	$31,948
Labor Costs						
6. Tractor operators (line 1 / (2 x line 3)[3] x $6/hr)	$3,162	$4,944	$7,428	$12,402	$18,528	$26,988
7. Total operating costs (line 4 + line 5 + line 6)	$8,025	$12,502	$18,814	$31,457	$47,265	$68,674
8. Total operating costs per cow	$80	$63	$63	$63	$63	$69

[1]Based on one hour per 16,000 cu. ft. waste.
[2]Hauling time (min.) per herd load: 100 cows–36 min.; 200–36; 300–42; 500–48; 750–54; 1,000–60.
[3]Time on line 3 is doubled because one person is at agitation pump site plus hauling time.
Source: Bennett et al. 1994.

TABLE A.15. Market value of delivered nutrients.

Basin State	Value of Nutrients *(Cents/Pound)*		
Iowa[1]	0.15	0.29	0.13
Indiana[2]	0.15	0.22	0.12
Missouri[3]	0.23	0.22	0.14

NOTE: Only nutrients used by crops hold value. Nutrients applied in excess of crop needs have zero value.
[1] Data are from Fleming et al. 1998.
[2] Data are from Doster et al. 1998.
[3] Data are from Bennett et al. 1994.

A.3 ECONOMICS AND POLICY CONSIDERATIONS

The hypoxia condition in the Gulf of Mexico is assumed to be a result of excessive nutrient loading from the Mississippi River. The objective of the Topic 6 team is to identify both sources of nutrients within the Mississippi watershed and policies that could assist in reducing nutrient loads. According to the assessment of this section, livestock manure is a significant source of nutrients. Because nitrogen has been chosen the nutrient target for causing hypoxia, policies should be designed for reducing nitrogen loads within the Mississippi Basin. For livestock, a policy that would require land application to be based on the phosphorus needs of crops could help control nutrient loadings from manure.

A few studies have addressed the impacts of such a policy on farmers, as well as its effectiveness in reducing water quality damages from animal nutrients. For example, Pratt et al. (1997) measured surface-water quality and changes in farm revenue for a variety of policies concerning dairy manure management in the upper North Bosque watershed in north central Texas. Fleming et al. (1998) measured the net benefit of a P standard for hog manure on continuous corn and corn–soybean rotations in Iowa.

When estimating the economic impact of the change in manure application policy, it is important to assess the net benefits and costs of manure disposal. The two most important factors that determine the net benefit of swine manure in crop production are the distance the manure has to be hauled and the nutrient content of the manure. The farther a given amount of manure must be hauled, the higher the cost. And, up to a point, the more nutrients that are delivered to a field, the greater the value (Fleming et al. 1998). It is accepted in the literature that manure has a nonzero value because of its nutrient content (Roka and Hoag 1996).

A.4 MANURE BENEFIT

The value of manure is determined by the cost of commercial nutrients. Therefore, the farmer benefits from applying manure to crops by reducing commercial fertilizer purchases. It is important to mention that those nutrients applied in excess of crop needs have zero value; hence, manure value is determined only by those nutrients used by crops. According to Fleming et al. (1998), when manure disposal is based on a phosphorus standard as opposed to nitrogen, the benefit of manure increases because all the potential value of applied nutrients is captured. For corn fields, the farmer will not benefit from reduced commercial fertilizer application because with a P standard the N requirement of corn will not be met, and the farmer will apply additional commercial N fertilizer.

A.4.1 Disposal Cost

Table A.12 shows that when basing manure disposal on the phosphorus needs of the crop, the acreage needed increases significantly. Table A.16 illustrates the variable costs for a pit and lagoon system in Iowa, which represents the costs incurred under different policies. Therefore, in most situations the marginal disposal cost increases due to the increase in distance needed to travel to dispose of the same amount of waste. This result was concluded in Pratt et al. (1997) as well as Fleming et al. (1998). However, Fleming et al. found that when the policy was applied to a corn–soybean rotation, marginal delivery costs actually decreased because under an N standard the farmer was not allowed to dispose of manure on soybean fields because soybeans do not require nitrogen supplements. Alternatively, soybeans do require phosphorus; hence, under a P standard the farmer is able to apply manure waste to soybean fields.

TABLE A.16. Average custom costs of delivering nutrients in Iowa swine manure to corn and soybeans.

Parameter Values	Pit Storage		Lagoon Storage	
	Nitrogen	Phosphorus	Nitrogen	Phosphorus
Unit Hauling Cost (cents/gal)	0.88	0.88	0.71	0.71
Unit Mileage Charge (cents/gal-mile)	0.34	0.34	0.28	0.28

Source: Fleming et al. 1998.

As herd size increases, the delivery cost of manure will increase to a point where the value of manure will no longer provide a significant reduction in the overall costs. It has been documented for the hog industry (Boland 1998) that as herd size increases there is a point where it becomes economical to store manure in lagoons. As herd size increases there is a greater need for minimizing nutrients from manure. This is due to limited land availability, delivery cost, and the need for meeting environmental constraints. A concern for farmers with the policy changing to a P standard is the increased land needed to dispose of the same amount of manure, which may lead them to switch to the lagoon sooner (at a lower number of head) than under the N standard. With the need for reducing nutrients, a method of storage that will minimize the amount of nutrients available may be the optimal solution. Therefore, farmers may switch to lagoon systems, as opposed to deep pits, to minimize nutrient availability and the land requirements needed for disposal.

Fleming et al. (1998) found that changing the manure disposal standard based on the crop's P needs benefited hog farmers. This is due to their analysis focusing on finding the optimal herd size based on the returns to manure nutrients, not on the return from the market value of hogs. Because the return on manure is significantly smaller than the market value of a hog, a farmer will tend not to make decisions based on manure value. Pratt et al. (1997) concluded for dairy farmers that a policy change to a P standard would reduce net returns under all policy options. We can conclude that in certain areas of livestock production, basing manure disposal on the phosphorus requirements of crops will generally reduce farm revenue.

A.5 NUTRIENTS

Not only is it important to understand the economic impacts of a policy change; the policy's effectiveness should be considered as well. For the hypoxia issue, this entails assessing whether the amount of excess nutrients is reduced. Nitrogen is the greatest concern, but phosphorus is also considered. Through the results of the AMANURE program from Table A.16, the amount of nutrients provided by manure was calculated. The program also determined the amount of nutrients overapplied (excess) and the amount supplied below crop needs (deficit). Table A.17 lists the amount of nutrients supplied by manure for each disposal method, crop, and nutrient target.

When the P standard is applied to corn, all nutrients are not applied in excess, and nitrogen disposal is actually reduced. Therefore, a P standard would be effective for reducing nutrient loadings from animal manure for corn fields. Alternatively, when the P standard is applied to soybean fields, nutrient loadings are different. Because soybeans need P, farmers can apply manure on soybean fields. Though this may not increase the amount of excess P, it does increase the amount of excess N, which was not in the system under an N standard because soybeans do not need nitrogen, so farmers were unable to spread manure on soybean fields. Also Pratt et al. (1997) found that switching to a P standard for coastal wheat rotation increases excess nitrogen. Therefore, if the objective for solving the hypoxia problem is to reduce the amount of nitrogen reaching the Gulf of Mexico, a P standard may not be an effective solution.

Table A.17 also shows the change in net benefit of the nutrients with the policy change. The net benefit is generated by first calculating the market value of nutrients used by the crop, and subtracting the value of nutrients lost through their overapplication. When application is based on the crop's phosphorus needs, less nutrient value is lost, and the net benefit is generally higher under both corn and soybean pit and lagoon systems.

A.6 CONCLUSION

The impact on water quality from animal manure nutrients has become an important issue in recent years. Nutrients from manure supplied within the Mississippi Basin are believed to contribute to the hypoxia problem in the Gulf of Mexico. The contribution of nitrogen within the system from animal agriculture has remained relatively constant over the past 40 years, although when the industry structure is examined, it is evident the concentration of animals has substantially increased. In these areas of high animal concentration, often more nutrients are supplied than what the environment can absorb, thereby resulting in water quality problems.

Under current land-application regulations, phosphorus tends to be applied at a rate greater than what the crops can absorb. As a result, it collects in soils and surface-water bodies, causing a variety of water quality problems. For the hypoxia issue, nitrogen has been the main focus for regulation. However, it is important to mention at this point that phosphorus was found to be “the forgotten nutrient” in the hypoxia in Chesapeake Bay (Blankenship 1997). Therefore, both nitrogen and phosphorus regulations may need to be considered.

As animal agriculture expands, it is important to ensure that there is enough land available for spreading manure. The methods applied within Chesapeake Bay for reducing the nutrient impacts from livestock include establishing nutrient management plans. The objective of these plans is to match the nutrient requirements of the crops for both nitrogen and phosphorus, thereby reducing runoff and leaching of excess nutrients (Alliance for Chesapeake Bay 1996a, 1996b; Blankenship 1998).

A policy alternative for potentially reducing nutrient loads from animal manure is to change land application to be based on the crop's phosphorus needs. While this policy is successful in reducing excess phosphorus in the system, it may not be effective in reducing nitrogen when not applied to corn fields. It is also found to reduce farm revenue.

Nutrients supplied by animal manure can be reduced to the Mississippi watershed by establishing manure management plans that match the amount of nutrients supplied to the crop's needs. In areas where nutrients are oversupplied, other disposal alternatives may need to be considered.

TABLE A.17. Comparing the amount of nutrients supplied by manure and their value for each nutrient target.

PARAMETERS	CORN (*111–125 bushels/acre*)				SOYBEANS (*41–50 bushels/acre*)			
	Nutrients Needed by Crops *(lbs./acre)*							
Nitrogen (N)	130				180*			
Phosphorus (P)	100				90			
Potassium (K)	180				210			
Storage Method	**Pit**		**Lagoon**		**Pit**		**Lagoon**	
Nutrient Target	**N**	**P**	**N**	**P**	**N**	**P**	**N**	**P**
	Nutrients Provided by Manure, Based on Nutrient Target (*pounds/acre*)							
Nitrogen (N)	130	75	130	78	180	68	180	70
Phosphorus (P)	173	100	167	100	240	90	232	90
Potassium (K)	168	97	212	127	233	87	294	114
	Nutrients Supplied by Manure That Were in Excess or Deficit of Crop Needs (*pounds/acre*)							
Nitrogen (N)	0	-55	0	-52	0**	68	0**	70
Phosphorus (P)	73	0	67	0	150	0	142	0
Potassium (K)	-12	-83	32	-53	23	-123	84	-96
	Value of Nutrients Used by Crops Based on Indiana Fertilizer Rates (*$/acre*)***							
	61.66	44.89	63.1	48.94	45	30.24	45	33.48
	Total Fertilizer Cost if Nutrient Needs Were Supplied by Commercial Fertilizer (*$/acre*)****							
	63.1	63.1	63.1	63.1	45	45	45	45
	Commercial Fertilizer Cost After the Application of Manure (*$/acre*)*****							
	1.44	18.21	0	14.16	62.76	0	68.32	11.52
	Total Value of Manure (*$/acre*)******							
	61.66	44.89	63.1	48.94	45	30.24	45	33.48

* Because soybeans can create their own nitrogen, they do not need nitrogen to produce. On the other hand, soybeans will use available nitrogen, such as that supplied by manure.

** Although 180 pounds of nitrogen were supplied by manure and taken up by the soybeans, a nitrogen credit is given because soybeans do not need the N, but will use what is available. Therefore, the N supplied by manure was neither over- nor underapplied,

*** This number is generated on the market value of nutrients: N = $0.15/lb; P = $0.22/lb; and K = $0.12/lb. These rates were then multiplied by the pounds of nutrients used by the crops. Nutrients supplied in excess are considered to have a zero value and do not contribute to the value of manure supplied to crops.

**** This is the cost (only the cost of the nutrients, not the application cost), the farmer would have to pay if the nutrients needed by crops were supplied by commercial nutrients only.

***** After crediting the nutrients from manure, the commercial fertilizer cost is shown below (nutrient cost only).

****** The value of manure is often referred to as the cost savings between what the farmer would have had to pay for commercial fertilizer minus what he actually had to pay after considering the nutrients supplied by manure. This value is the same as the value of nutrients supplied times the market rate for nutrients.

NOTES: According to the current regulations, manure is to be supplied based on the nitrogen needs of the crops. Since soybeans do not need nitrogen, farmers may not apply manure to their soybean fields.

The phosphorus needs for each crop are based on the assumption that the soil test P levels are 0–5 ppm. In addition, the K needs are based on the assumption that soil test K levels are 0–40 ppm. These nutrient needs numbers can be obtained from any fertilizer recommendations table.

APPENDIX B

Atmospheric Deposition of Nitrogen

B.1 BENEFITS AND COSTS OF REDUCING DIFFERENT SOURCES OF ATMOSPHERIC DEPOSITION OF NITROGEN

As it has been broadly explained in previous sections, nitrogen is responsible for several varieties of adverse consequences to the health of estuaries and coastal waters. Traditional sources of nitrogen pollution that have been extensively studied are direct discharges from sewage treatment and agricultural runoff.

Recently, atmospheric deposition of nitrogen has gained attention as another important source of nitrogen pollution. For example, the latest studies in the Chesapeake Bay estuary established that atmospheric nitrogen deposition in the watershed is responsible for 27% of the nitrogen load in the bay, suggesting that atmospheric deposition cannot be excluded from any analysis of the nitrogen pollution issue.

B.2 RECENT FINDINGS ON SOURCES AND LOADINGS

B.2.1 Characteristics of Atmospheric Nitrogen Sources

Most nitrogen compounds in the atmosphere fail into three categories: reactive nitrogen, reduced nitrogen, and organic nitrogen. Reactive nitrogen is the largest contributor to atmospheric deposition—40–60%; reduced nitrogen follows, with a contribution of 20–40%; and finally organic nitrogen, which contributes 0–20%.

Reactive nitrogen compounds are primarily oxides of nitrogen (NO_x), which are produced during the combustion of fossil fuels. Dominant sources of NO_x include emissions from motor vehicles and power-generation plants. The NO_x reacts in the atmosphere with other gases, initiating a cycle that eventually turns into ozone (O_3) and nitric acid (NHO_3), which is easily and quickly deposited on the surface and runs into streams.

Reduced nitrogen compounds are ammonia (NH_3) and ammonium (NH_4+). Once again, the most important source of these compounds is fossil fuel combustion. These compounds are highly reactive and have a short life in the atmosphere. They quickly deposit in the areas near their source, react with other components, and convert into nitrate salts, which finally end up in streams.

Organic nitrogen is the result of NO_x reaction with certain hydrocarbons during combustion. This type of nitrogen pollutant has been currently studied to some extent; however, its contribution to nitrogen pollution in coastal waters is still a matter of research.

B.2.2 Recent Findings on Sources and Loadings

In the last few years several efforts have been taken to calculate watershed loads from atmospheric deposition. A recent analysis by the National Atmospheric Deposition Program (currently undergoing final review), entitled "Atmospheric Deposition Estimates of Nitrogen to the Atlantic and Gulf Coast of the United States," provides some highlights for average annual deposition:

- nitrate loading ranged from 1.26 to 4.56 kg/ha;
- ammonium nitrogen ranged from 1.01 to 2.50 kg/ha; and
- organic(inorganic) nitrogen ranged from 2.78 to 6.71 kg/ha.

According to Scott Dinnel's (1995) *Estimates of Atmospheric Deposition to the Mississippi Watershed* and EPA's (1998b) *The Regional NO_x SIP Call and Reduced Atmospheric Deposition of Nitrogen*, atmospheric deposition loads from the Mississippi watershed are 17% of the total nitrogen load to the Gulf. Another study by Baker (1996) has estimated that atmospheric deposition contributes about 23%. of the total nitrogen load in the Gulf coast estuaries.

Considering the importance that most recent research has given to atmospheric nitrogen deposition as a contributor of nitrogen pollution into coastal waters, control of atmospheric nitrogen sources has a significant potential for reducing total nitrogen loadings. As noted above, atmospheric nitrogen is largely produced from the combustion of fossil fuels by automobiles and power-generation plants. Mobile sources of atmospheric nitrogen (motor vehicles) contribute 30–40% of the total atmospheric nitrogen deposition in Chesapeake Bay, and utility and non-utility point sources contribute 30–50% of the total deposition (Dennis, in press).

In a few words, atmospheric loads from fossil fuel combustion represent a large proportion of the total nitrogen deposited to surface waters according to research from several organizations in recent years. Reducing emissions of atmospheric nitrogen from the burning of fossil fuels has a great potential for significantly decreasing nitrogen loadings and, therefore, for improving the health and ecology of estuaries.

Tampa Bay and Long Island Sound are focusing on controlling nutrient runoff from urban nonpoint sources, which is the most expensive nitrogen to control. Albemarle/Pamlico Sound and Chesapeake Bay have more diverse conditions, including urban and agricultural sources and controls. The difference between the latter two is that the first includes a larger proportion of agricultural sources and controls than the second. Given the size and diversity of the Mississippi Basin, the cost of additional controls of nonatmospheric nitrogen would be between the values obtained for Albemarle/Pamlico Sound and Chesapeake Bay (Table B.1).

TABLE B.1. Cost of reducing nitrogen load from nonatmospheric sources.

	Estuary	Nitrogen Load Reduction	Cost of Reduction		
		1,000 kg/yr	Mill. $/yr	$/kg	$/lb
1	Albemarle/Pamlico Sound	1,870	136.00	72.71	32.98
2	Cape Cod Bay	570	78.00	136.84	62.00
3	Chesapeake Bay	4,010	193.00	48.12	21.83
4	Delaware Bay	700	96.00	137.14	62.00
5	Delaware Inland Bays	80	11.00	137.50	62.00
6	Gardiners Bay	180	25.00	138.89	62.00
7	Hudson Bay	650	89.00	136.92	62.00
8	Long Island Bay	840	196.00	233.33	105.84
9	Massachusetts Bay	190	26.00	136.84	62.00
10	Narragansett Bay	180	25.00	138.89	62.00
11	Sarasota Bay	10	1.00	100.00	62.00
12	Tampa Bay	70	13.62	194.53	88.24

B.3 COST OF REDUCING THE AMOUNT OF ATMOSPHERIC NITROGEN DEPOSITION

In reviewing cost of reducing atmospheric nitrogen deposition, we faced the same problem as studying the benefits. Unfortunately, this area is just starting to be studied and very few studies have been undertaken. So far, there are no estimations for the Mississippi Basin, and the best cost analysis found to date is *Atmospheric Nitrogen Deposition Loadings to the Chesapeake Bay: An Initial Analysis of the Cost-Effectiveness of Control Options* (Pechan 1997). This section describes the estimates obtained for Chesapeake Bay and suggests how to use that information to obtain an adequate approximation of the cost for the Mississippi Basin.

This report presents the cost of reducing NO_x emissions from the most important sources—utilities, motor vehicles, and other nonutility point sources that burn large amounts of fuel fossils. Table B.2 summarizes the cost of reducing nitrogen loads from atmospheric sources. Figures B.1 and B.2 show the cost of reducing nitrogen loads per pound of nitrogen from utility and mobile sources as a function of the total reduction in nitrogen emission.

B.4 TRADE-OFFS BETWEEN POINT AND NONPOINT SOURCES IN REDUCING ATMOSPHERIC NITROGEN LOADINGS

Since this area of research on economic impacts is just starting to develop, not many studies have been conducted. One of the most recent appraisals of the economic impacts from reducing atmospheric nitrogen is EPA's *Benefits of Reducing Deposition of Atmospheric Nitrogen in Estuarine and Coastal Waters* (1997).

TABLE B.2. Nitrogen load reduction from atmospheric sources and cost by source.

REGION	SOURCES OF EMISSIONS			
	Utilities	Motor Vehicles	Non-utility Point Sources	Total
	NO_x Emission Reductions (tons/year)			
Mid-Atlantic States				
Delaware	11.4 (80%)	2.0 (14%)	0.9 (6%)	14.3
District of Columbia	0.2 (13%)	1.2 (80%)	0.1 (7%)	1.5
Maryland	47.0 (75%)	13.6 (22%)	2.3 (4%)	62.9
New Jersey	28.7(52%)	20.6 (37%)	6.2 (11%)	55.5
New York	63.7 (58%)	36.6 (33%)	10.4 (9%)	111.0
Pennsylvania	178.0 (86%)	24.1 (12%)	5.4 (3%)	208.0
Northern Virginia	6.9 (61%)	4.4 (39%)	0 (0%)	11.3
Other States				
Kentucky	169.0 (98%)	3.9 (2%)	0.3 (0%)	173.0
North Carolina	92.4 (82%)	7.6 (7%)	13.3 (12%)	113.0
Ohio	248.0 (90%)	10.9 (4%)	17.8 (6%)	277.0
Tennessee	104.0 (76%)	5.8 (4%)	26.8 (20%)	136.0
Virginia	45.9 (82%)	6.0 (11%)	3.9 (7%)	55.8
West Virginia	156.0 (93%)	1.7 (1%)	9.4 (6%)	167.0
	Cost of NO_x Emission Reduction *($/ton/year)*			
Mid-Atlantic States				
Delaware	1,800	3,200	9,800	
District of Columbia	2,100	2,800	3,100	
Maryland	1,300	2,900	8,200	
New Jersey	1,900	2,700	710	
New York	1,900	2,600	6,700	
Pennsylvania	1,200	3,200	9,800	
Northern Virginia	2,000	2,700	0	
Other States				
Kentucky	1,100	7,600	9,500	
North Carolina	1,100	7,600	5,700	
Ohio	1,200	7,400	6,100	
Tennessee	1,100	7,800	4,900	
Virginia	1,000	7,600	5,800	
West Virginia	1,000	7,500	6,300	

The basic approach followed by this EPA study is to evaluate benefits from reducing atmospheric nitrogen loadings as the opportunity cost of reducing nitrogen from other nonpoint sources that would be needed in order to meet and maintain water quality goals in the estuaries examined. Twelve estuaries were selected for these studies: 10 are along the Atlantic coast and two—Tampa Bay and Sarasota Bay—are along the Gulf coast. Since sufficient data were not available for all the estuaries, detailed costs were obtained for only four estuaries: Tampa Bay, Albemarle/Pamlico Sound, Chesapeake Bay, and Long Island Sound. Then an average was obtained from these four estuaries to obtain a cost for the other eight estuaries. We concentrated on the four watersheds with detailed information.

The cost of nonpoint-source controls ranged from \$0.61 to \$45.27 per pound of nitrogen load for agricultural best management practices and from \$35.00 to \$142.64 per pound of nitrogen load for urban nonpoint-source controls. Table B.1 shows the cost estimates for the different estuaries. Higher cost per unit of nitrogen is associated with smaller, more urban estuaries, which typically have fewer technical and financial options available. For the eight other estuaries, the mean cost per pound of nitrogen from the four case estuaries was estimated to be \$62.00.

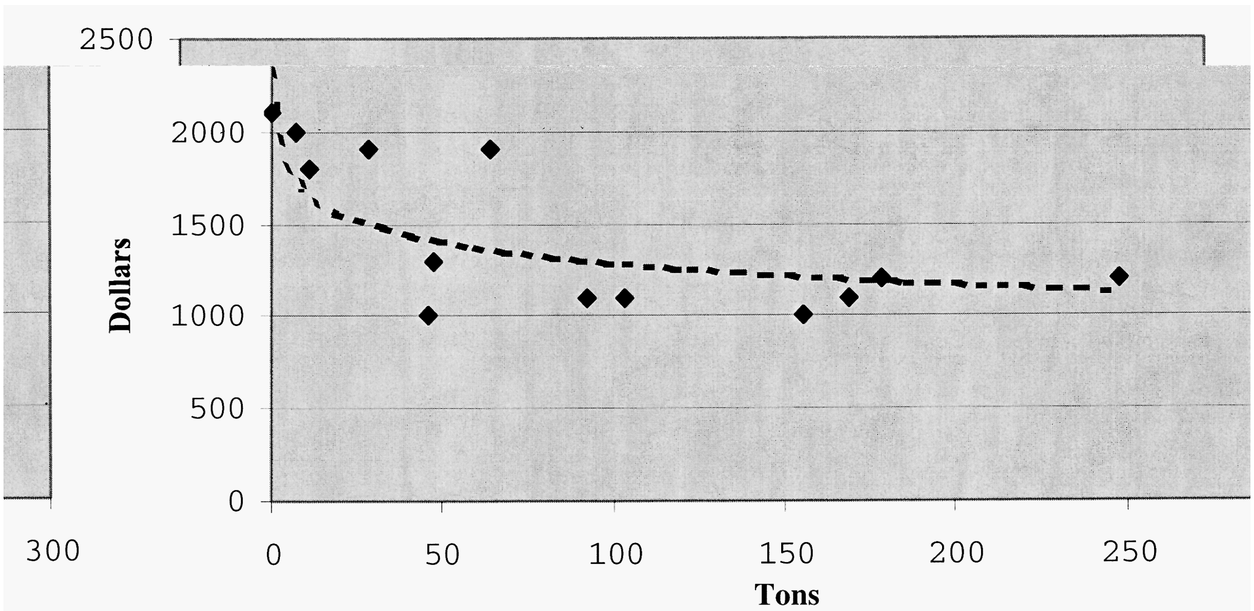

FIGURE B.1. Utility source cost per ton of NO_x emission reduction.

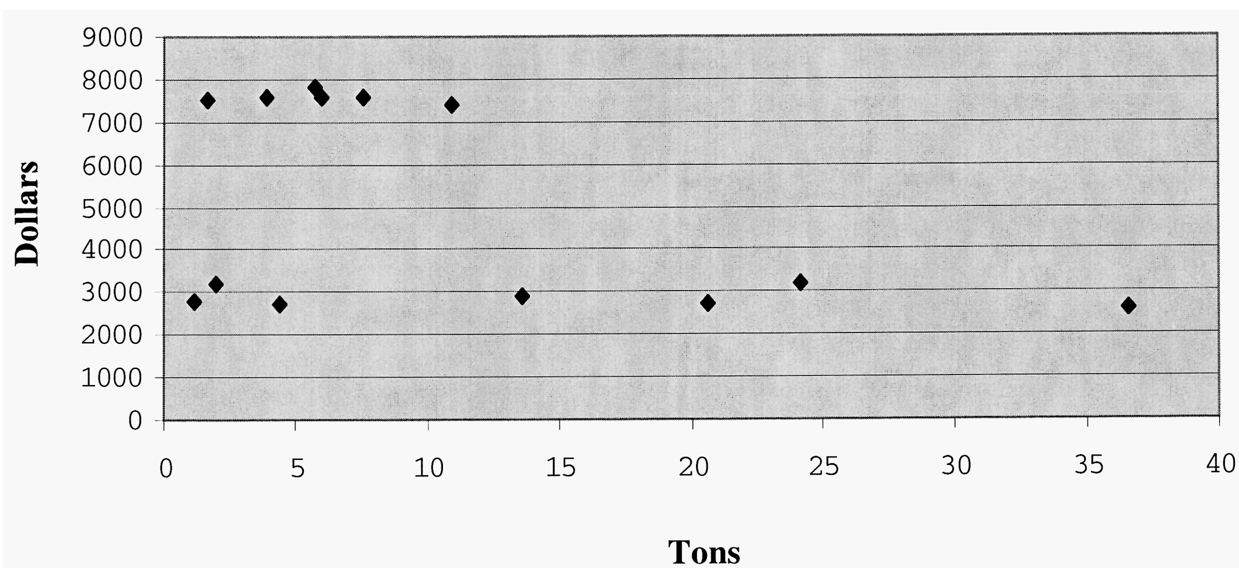

FIGURE B.2. Motor vehicle source cost per ton of NO_x emission reduction.

Finally, we estimated the cost per pound of nitrogen loaded into Chesapeake Bay by calculating the percentage of the deposition that eventually will end up in the bay. The percentage contribution changes by region: the closer to the ocean, the higher the proportion of nitrogen deposited on the ground that ends up in the bay. Table B.3 shows our results.

TABLE B.3. Nitrogen load reduction and cost.

Source/ State	NO_x Reduction	Nitrogen Load Reduction	Total Annual Cost	Cost-Effectiveness		
	Thousands of Tons	Thousands of Pounds	Millions of Dollars	\$/ton NO_x Emission	\$/lb NO_x Emission	\$/lb N Load in Chesapeake Bay
Utility Sources						
Maryland	47.0	1,610	62.7	1,334.04	0.67	38.94
Pennsylvania	178.2	3,510	214.0	1,200.90	0.60	60.97
Virginia	52.8	1,990	57.9	1,096.59	0.55	29.10
West Virginia	155.5	2,240	157.5	1,012.86	0.51	70.31
Kentucky	169.1	7,60	192.3	1,137.20	0.57	253.03
Mobile Sources						
Maryland	13.6	410	39.0	2,867.65	1.43	95.12
Pennsylvania	24.1	470	76.5	3,174.27	1.59	162.77
Virginia	4.4	90	11.9	2,704.55	1.35	132.22
West Virginia	10.4	220	58.4	5,615.38	2.81	265.45

APPENDIX C

Overview of the EPIC Model

C.1 THE EROSION PRODUCTIVITY IMPACT CALCULATOR MODEL

The Erosion Productivity Impact Calculator (or EPIC) model was developed in the early 1980s by the U.S. Department of Agriculture's (USDA's) Agricultural Research Service to assess the long-term impacts of soil erosion on crop productivity. EPIC is a daily time-step mathematical simulation model that integrates numerous physical and biological components, including crop growth, hydrology, weather simulation, nutrient cycling, crop management practices, erosion–sedimentation, economic accounting, soil temperature, and plant environment control (drainage, irrigation, fertilization, etc.). Unlike many hydrologic models that focus solely on the physical processes, EPIC explicitly models the daily interaction of crop growth, soil, weather, and nutrient cycling. The model is also capable of simulating multiple cropping systems and management practices over long periods of time (50–4,000 years).

Because soil erosion can take several decades to affect crop yields, the EPIC model was designed to attain four goals: (1) develop a realistic, physically based model of erosion with readily available input data; (2) simulate processes over long time horizons; (3) produce valid results over a wide range of crops, soils, and climates; and (4) provide a computationally efficient model. The current PC and mainframe versions of the model accomplish these goals.

C.2 OPERATIONAL OVERVIEW

EPIC is characterized as a lumped-parameter model because the drainage area considered, usually around one hectare, is assumed to be spatially homogeneous on the surface. It assumes the area simulated is a single soil type with constant slope and surface roughness throughout. However, the model does consider vertical variation in soil properties associated with a given soil profile.

The first step in using EPIC involves specifying simulation input parameters that include general information on the size, slope, and elevation of the area; weather and soil data; and crop management information. The weather data can either be entered from actual daily observations or simulated from long-term distributions. Input data for daily weather simulation include monthly distributional parameters for maximum and minimum temperature, precipitation, Markov probabilities of wet/dry states, humidity, solar radiation, and wind velocity and direction. Soil input data include 21 parameters for each layer and up to 10 layers per soil type. The crop management information entails the sequencing of crops, timing of planting and tillage, fertilizer amounts and timing, irrigation if required, and harvest operations.

On a daily simulation basis, weather data are used as input into several sub-routines that track surface and subsurface hydrology, evaporation, changes in soil properties, crop growth, nutrient cycling, evapotranspiration, crop stress, and erosion. EPIC uses a generalized plant growth model with crop-specific coefficients to estimate daily plant growth. Daily plant growth is partitioned between roots, above-ground biomass, and crop yield. Estimated daily crop growth can be limited by one or more daily stress factors, including water, nutrients (N and P), temperature, and aeration. All daily values are retained, and the model goes on to simulate the next day. The daily time-step simulation continues by completing the annual loop, and then the multiple crop-rotation loop if it entails multi-year crop sequencing, until the entire evaluation period has been covered.

The EPIC model can also be used to estimate crop yield and environmental parameter distributions by cropping systems. These distributions can be derived by changing the actual weather sequence or changing a seed number for the random weather generator. This feature of EPIC is important if one is interested in evaluating crop yield risk by management or fertilizer practice, as well as determining the range and probability of nutrient and pesticide loadings.

EPIC simulations have been performed on over 160 test sites in the continental U.S. and Hawaii. These tests have shown that EPIC produces valid results over a variety of climatic conditions, soil characteristics, and management practices (Williams et al. 1984). In addition, Foltz (1991) has shown that agronomic experts could not differentiate between EPIC-simulated crop yields and actual experimental plot yields over a four-year period in Indiana.

C.3 MODEL APPLICATIONS

While the EPIC model has been used extensively to evaluate the impact of soil erosion on crop productivity, more recent applications of the model range from simulating the movement of pesticides and nutrients to large-scale assessment of global climate change. The integrated modeling structure of EPIC and readily available data make the model conducive to evaluate a broad class of research issues. In fact, the model's name has recently been revised to the "Environmental Policy Integrated Climate" model to reflect this flexibility.

One area where the EPIC model has been used is to provide per-acre/hectare estimates by cropping systems of sediment, nutrient, and pesticide loads. These estimates are then often used as technical coefficients in farm-level or regional mathematical programming models. Foltz et al. (1995) used EPIC to stimulate crop yields, organic nitrogen loss in runoff and percolate nitrogen for crop-rotation alternatives in the Midwest. Economic and environmental parameters from EPIC provided input for a multi-attribute assessment of cropping system selection.

Specific to water quality research with EPIC, Mapp et al. (1994) used EPIC-PST to assess the economic and environmental impacts of limiting nitrogen fertilizers to protect water quality. Likewise, Helfand and House (1995) applied EPIC to California lettuce production to evaluate taxation versus regulation to reduce agricultural nonpoint-source pollution. Finally, Randhir and Lee (1997) used EPIC-WQ to simulate crop growth and pollutant loads for alternative cropping systems to provide input data for a multi-year regional risk-programming model. Results from this study show that nitrogen and pesticide restrictions involve not only economic and environmental trade-offs but also trade-offs among various nonpoint-source pollutants.

C.4 STRENGTHS OF EPIC

One of the primary strengths of EPIC is its integration of physical and biological processes in a systematic modeling framework. EPIC can simulate the growth of over 20 crops ranging from corn, soybeans, and cotton to legume crops and trees. Over 600 soil parameter files and over 100 long-term weather station data input files are available for application in the U.S.

The model provides a wide range of biological and physical parameter outputs. On the biological side, these output variables include crop yield, crop growth, stress days, nutrient uptake, and crop residue. On the physical side, the model provides estimates of nutrient runoff (soluble and sediment attached), nutrient losses to subsurface flow, and leachate. These parameters can be reported as loadings (i.e., pounds per acre/hectare) or as concentrations (i.e., ppm). Other physical parameters include runoff volumes, pesticide runoff and leachate, and soil erosion. Specific to soil quality, the model can simulate changes in soil properties over time resulting from different crop and management sequences. These properties include soil organic matter, cation exchange capacity, pH, field capacity, and changes in soil layer thickness.

From a policy perspective, output from EPIC can be used in analytic models to quantify the economic and environmental impacts of crop production and management changes. Simulation results derived under random weather conditions can be used to generate distributional information on crop yields, pollutant levels, and soil loss. These estimates are often required for risk assessment analysis.

C.5 LIMITATIONS OF EPIC

As with any model, the application of EPIC has inherent limitations. One of the first is the direct use of EPIC to simulate agricultural nonpoint-source pollution on a field or watershed scale. EPIC is a lumped-parameter model, which means that all hydrologic and soil parameters are uniform within the simulated region. Therefore, EPIC does not account for spatial interdependence of pollutant loadings that can occur within a field or watershed. Because EPIC is a lumped-parameter model, it is difficult to validate nutrient and pesticide runoff from watershed-level, water quality monitored data. For example, tile-drainage systems affect measured nutrient loads at the watershed outlet but are difficult to account for in the EPIC framework.

A second limitation of EPIC concerns the large number of input variables required to run the model (over 200) and the large number of output variables it produces (over 125), which make model validation difficult. Most applications of EPIC validate and calibrate the model to crop yield. Few studies have validated EPIC predictions of key environmental parameters, such as nutrient or pesticide runoff and leachate on actual plot data. Many of the environmental estimates are highly sensitive to key input parameters.

A final limitation of the model is the level of simplification of many of its sub-routines. For example, EPIC's crop growth component is a simple daily biomass-accumulation model. Daily biomass is partitioned between roots, above-ground biomass, and crop yield. This simplification can cause problems if one wishes to model crop responses, for example, to climate change, because EPIC ignores important biotic processes, such as stomatic resistance. EPIC also does not explicitly consider pest damage, plant diseases, weed pressure, or soil microorganisms. Likewise, one may have difficulty in modeling new varieties or genetically engineered crops. The model contains over 25 crop-specific parameters. In modeling a new crop, knowing which parameter(s) to adjust and by how much becomes more of an art than a science.

C.6 CONCLUSION

Despite EPIC's limitations, the model is still well suited to evaluate the long-term impacts of soil erosion on crop productivity. Over the past decade the model has been extended to simulate pesticide loadings from crop production. Applications of EPIC range from hybrid tree production to animal waste disposal impacts on crop yields and pollutant loadings.

In recent years, EPIC has been modified to address some of the spatial concerns. One modification, called APEX, is a hill slope version of EPIC. APEX can simulate three different sloped/cropped areas and include hydrologic routing between each area. This model extension can be useful to evaluate the water quality benefits of buffer strips or riparian buffers to reduce agricultural nonpoint-source pollution.

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