Chapter 4
Scientific Basis for Goals and Management Options

4.1 Adaptive Management

Adaptive management offers a way to address the pressing need to take steps to manage for factors affecting hypoxia in the NGOM in the face of uncertainties. The authors of a recent study undertaken by the National Research Council of the National Academy of Sciences identified six elements of adaptive management that are directly relevant to goal setting and research needs (National Research Council, 2004): (1) resources of concern are clearly defined; (2) conceptual models are developed during planning and assessment; (3) management questions are formulated as testable hypotheses to guide inquiry; (4) management actions are treated like experiments that test hypotheses to answer questions and provide future management guidance; (5) ongoing monitoring and evaluation is necessary to improve accuracy and completeness of knowledge; and (6) management actions are revised with new cycles of learning.

Perhaps the most important "take-home" lesson from their work is contained in the following statement:

> Adaptive management does not postpone actions until "enough" is known about a managed ecosystem (Lee, 1999), but rather is designed to support action in the face of the limitations of scientific knowledge and the complexities and stochastic behavior of large ecosystems (Holling, 1978). Adaptive management aims to enhance scientific knowledge and thereby reduce uncertainties. Such uncertainties may stem from natural variability and stochastic behavior of ecosystems and the interpretation of incomplete data (Parma et al., 1998; Regan et al., 2002), as well as social and economic changes and events (e.g., demographic shifts, changes in prices and consumer demands) that affect natural resources systems.

Thus adaptive management provides an appropriate way for decision makers to deal with the uncertainties inherent in the environmental repercussions of prescribed actions and their influences on hypoxia.

Adaptive management can be conducted at the several management scales that occur in the NGOM and MARB. On the basin scale, adaptive management requires measurements of both nutrient loadings and hypoxia extent (area). Although it will not be possible to relate these changes to specific changes in the basin, these data will provide better understanding of the relationships between nutrients and

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hypoxia. On smaller scales, specific management actions can be treated as experiments that test hypotheses, answer questions, and thus provide future management guidance at that scale (for example, small watersheds).

The adaptive management approach requires that conceptual models are developed and used and that relevant data are collected and analyzed to improve understanding of the implications of alternative practices (e.g., Ogden et al., 2005). To help illustrate what is meant by a conceptual model, the Study Group has developed a diagram that shows major factors that affect hypoxia in the NGOM (Fig. 4.1). The corresponding conceptual model would estimate the relative contribution of each influence. Those estimates could serve as hypotheses of relative effects, and the diagram could illustrate hypothesized interactions and feedbacks. Such a conceptual model organizes how adaptive management research is conducted in a framework where the testing of hypotheses and the new knowledge gained is then used to drive management adaptations, new hypotheses, and the collection of new data on end points. Unlike the traditional model of hypothesis-driven research, adaptive management implies coordination with stakeholders and consideration of the economic and technological limitations on management. Unlike traditional demonstration projects, adaptive management implies an understanding that complex problems will require iterative solutions that will only be possible through generation of new knowledge as successive approximations to problem solving are attempted.

Successful implementation of the adaptive management process is occurring in the Grand Canyon (Meretsky et al., 2000) and the Everglades (Sklar et al., 2005). In addition, steps toward adaptive management are being examined in the Upper Mississippi River basin (O'Donnell and Galat, 2008). That work documents the need for greater collaboration between scientists and management agencies to plan, design, and monitor river enhancement programs. Problems exist in setting quantifiable success criteria, developing appropriate monitoring designs, and disseminating

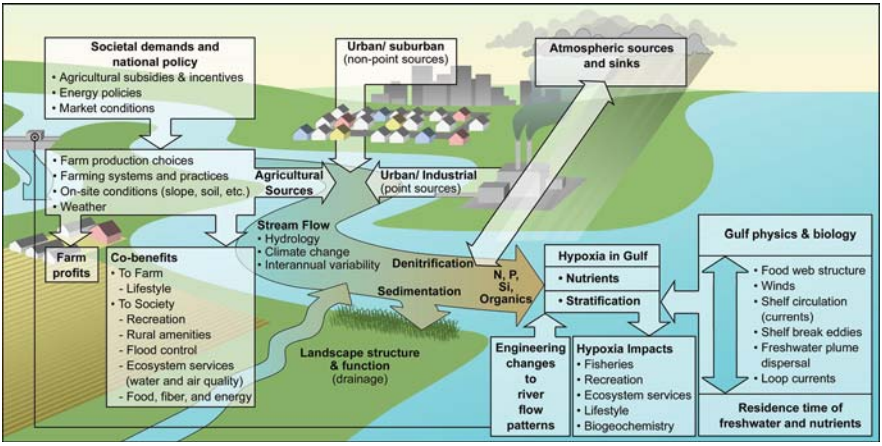

Fig. 4.1 A conceptual framework for hypoxia in the northern Gulf of Mexico

information. The Study Group expects similar difficulties in implementing adaptive management to occur in the MARB.

There needs to be a better understanding of the spatial and temporal aspects of basin-level responses to management practices and also a focus on other scales at which response can occur in a more timely fashion (Nassauer et al., 2007). Yet observations of a basin-level response to practices cannot be expected for some time, which calls for management and evaluation to be focused on a subbasin scale. Therefore it is important to obtain information at a scale where practices can be broadly and appropriately applied and where results are "meaningful and interpretable." The relevant scale would likely be at smaller subwatershed scales, where local water quality and quantity benefits may become evident more quickly. Furthermore, the demonstration of adaptive management within a small sub-watershed may enhance practice adoption at other locations. Thus conceptual models need to be developed for this scale of resolution as well. Focus at the small watershed scale will also provide local water quality and quantity benefits. The results from small watershed studies must be able to be extrapolated to other small watersheds in the subbasin and, preferably, the entire MARB, if they are to be useful in reducing hypoxia in the NGOM.

Experiments that could be applied at small watersheds to help to improve understanding of the effects of different practices have the following characteristics:

- Practices applied on the small watersheds should conform to accepted practice standards or make specific modifications of practices that can be implemented in new standards.
- Monitoring should be at appropriate intensities (time and space) to determine effects of practices on water quality and quantity.
- Monitoring should also measure co-benefits, including carbon sequestration, wildlife habitat, flood control, etc.
- Practices should be applied in suites or systems, and components should be monitored to determine effects of component practices.
- Changes in hydrology and crop productivity must be measured in addition to changes in water quality. Even at the small-scale, too many studies have focused just on nutrient concentrations in outflow water and neglected hydrologic or productivity changes.
- All components of the cost of adopting and maintaining these practices should be measured and monitored. Such costs include direct equipment and structural costs, yield effects, changes in management time, changes in risk, and other costs.
- These studies should be designed to improve our understanding at local, medium, and broad basin scale. Thus the experiments should be designed so that they can feed into conceptual models that operate at different scales.
- Within practical limits, studies should be part of an adaptive management research strategy for the MARB to optimize the efficiency of research investments and to assure that results are coordinated, complimentary, and consistent.

Integrated modeling and monitoring play an important role in adaptive management. The cornerstone of adaptive management is the concept of learning about the impacts of actions and using that new understanding to guide future actions. Models can assist that learning by being used to evaluate impacts and uncertainties of proposed actions, such as targeted practices and locations or proposed policies, on both MARB and NGOM responses. In addition, monitoring must also be part of an adaptive management strategy in order to verify that the actions are addressing the stated goals or to test hypotheses. Monitoring is needed to improve the next generation of models and model assessments and to eventually verify that projected changes occur.

Adaptive management is also important to building infrastructure and to strategic planning and policy development of mechanisms of conservation practice implementation. For example, adaptive management can be used to evaluate if incentive-based programs are effective at bringing about changes in conservation practice acceptance and adoption at a local or small watershed level. At a basin level, other programs might be needed to facilitate adaptation of strategies and policies, and there must be constant feedback among all vested parties. As the scale of system increases (i.e., from a small watershed to the entire MARB), the complexity of adaptive management increases dramatically.

Key Findings and Recommendations

Adaptive management can be used at several scales of resolution in the NGOM and MARB to provide a framework under which management activities can occur while monitoring and modeling the outcomes in order to provide information so that subsequent management can be improved. Therefore, the Study Group offers these recommendations.

- An adaptive management approach should be adopted to evaluate the success of reaching goals and for testing hypotheses (at the relevant scale).
- Conceptual models should be developed at appropriate scales of resolution to frame the adaptive management process in addressing factors affecting hypoxia in the NGOM.
- Both the use of quantitative models and the collection of data should be conducted within an adaptive management framework and at appropriate management scales so that the information gained from models and data are related to the critical questions about managing and understanding the system.
- Management actions should be designed as experiments within the context of evolving conceptual understanding of the system. The repercussions of management actions need to be monitored so the outcomes can be used to enhance learning and thus to improve future management actions.

4.2 Setting Targets for Nitrogen and Phosphorus Reduction

To reduce hypoxia in the bottom waters of the NGOM, the *Integrated Assessment* set a target that *N* loading should be reduced by 30% in order to shrink the 5-year running average size of the hypoxic zone to below 5,000 km^2 (1,930 mi^2) by 2015. This reduction is significantly less than the three- to five-fold increase in *N* loading to the Gulf of Mexico due to human activity during the 20th century, and particularly in the past 30–50 years (Boyer and Howarth, 2008; Goolsby et al., 2001). Since the *Integrated Assessment*, a number of modeling efforts have provided a better depiction of how the area of hypoxia may respond to reduced N loading. The three available models were compared by Scavia et al. (2004), who concluded from these models that the 30% reduction in N is probably not sufficient to reach the goal of a hypoxia area of 5,000 km^2 or less (Scavia et al., 2004). The consensus from these models is that N loads probably need to be reduced by 40–45% to reach the hypoxia reduction goal. In addition, a number of studies suggest that the consequences of climate change need to be considered, and this may require an N load reduction on the order of 50–60% to meet the original *Integrated Assessment* goal for hypoxic area (Donner and Scavia, 2007; Justić et al., 2003b). However, predicting the consequences of climate change on nutrient fluxes and hypoxia remains a very uncertain business (Howarth et al., 2006). The Study Group finds that the consensus of models reported by Scavia et al. (2004) and the new model of Scavia and Donnelly (2007), which uses the latest available load estimates from the USGS, supports a target of reducing the 5-year running average of N loadings by at least 45%. This target should be reassessed as more monitoring data are obtained, current models are refined, and new models are developed.

Only recently has new evidence emerged for the need to control P inputs as well as N in the NGOM. Work by Sylvan et al. (2006) has shown P to be the limiting nutrient during periods of maximum primary production in the near-shore NGOM high-productivity zone. Because previous attention has focused on N, there has been limited effort to model the effects of P on hypoxic area. Scavia and Donnelly (2007) used the previously developed and calibrated model (Scavia et al., 2004) to evaluate both the effects of new USGS load estimates and to assess the potential for P to control hypoxia dynamics under current and historical conditions. Confirming the results of Sylvan et al. (2006), Scavia and Donnelly found that P could have become limiting in some areas and times because of the relative increase in N loads during the 1970s and 1980s. While they concluded that P did frequently control hypoxia in near-field zone of NGOM, they noted that a P-only strategy would likely reduce production in the near-field but possibly increase production in downfield N-controlled areas of NGOM. Their work, using the new USGS load estimates, reinforced the need for a dual nutrient strategy combining a 45% reduction in N with a 40–50% reduction in the 5-year running average of P loading. While the far-field effects could possibly be reduced through an N-only strategy, they suggested that a prudent approach would be to reduce both N and P, simultaneously. They also noted that an N-and P-reduction strategy would not only reduce hypoxia in the NGOM but would also help to remove P-induced Clean Water Act impairments in the MARB. Based on this recent modeling work, the Study Group finds that a comparable P reduction is needed, again based on 5-year running average fluxes. As with the N target, this P target should be reassessed over time as more monitoring information is gained and new models are developed.

The CENR report and Scavia et al. (2004) made recommendations on an N reduction target with reference to average fluxes for 1980–1996. These fluxes were calculated using different methods (see Section 3.1) than in this book, but the N reduction target proposed recently by Scavia and Donnelly (2007) used a combination of the newer USGS 5-year LOADEST and composite estimates since 1980. In this book we only use the 5-year LOADEST results, since the composite estimates are incomplete; however, they are very similar to each other (again, see Section 3.1).

During the past 5 years of record, annual water flux to the NGOM has declined by 5.8%, whereas nitrate-N and TKN have declined even more, leading to a total annual N reduction of about 21% (Table 4.1). Considering the original reduction target of a 30% reduction in total N, it would seem that substantial progress was made beyond the reduction that would occur from less flow alone. However, the largest reduction was in TKN, with a large part of this decrease from the Missouri River (discussed in Section 3.1). For the important spring flux of N, there was little reduction in nitrate-N beyond the reduced water flow (–11 and –12.4% declines in water and nitrate-N flux, respectively). Again, TKN was greatly reduced (–31.5%) during spring flows, leading to most of the decline in total N (–19.2%), beyond the reduction in water

Table 4.1 Annual and spring (sum of April, May, June) average flow and N and P fluxes for the MARB for the 1980–1996 reference period compared to the most recent 5-year period (2001–2005). Load reductions in mass of N or P also shown

	Flow (million m^3(water) or million metric tons)			Flux (million metric tons)	
	1980–1996 flux	2001–2005 flux	Change (%)	45% reduction N target flux	45% reduction P target flux
Annual					
Water	692,500	652,500	–5.8		
Nitrate-N	0.96	0.81	–15.4	0.53	
TKN	0.61	0.43	–30.0	0.34	
Total N	1.58	1.24	–21.1	0.87	
Total P	0.137	0.154	+12.2		0.075
Spring					
Water	236,800	210,600	–11.0		
Nitrate-N	0.38	0.33	–12.4	0.21	
TKN	0.21	0.14	–31.5	0.12	
Total N	0.59	0.48	–19.2	0.32	
Total P	0.046	0.050	+9.5		0.025

flux. This suggests that during the important high-flow spring period (April, May, June), reductions in nitrate-N flux to the NGOM have not occurred under management systems and programs now in place since the most recent report. However, the annual nitrate-N reduction indicates that the tile-drained corn and soybean systems in the Upper Mississippi and Ohio River subbasins seem responsive on an annual basis to the recent reductions in net N inputs, as discussed in Section 3.2. Whether spring nitrate-N loads will respond to these changes in NANI is uncertain at this time.

For total P flux, both annually and during the spring, there were increases of 12.2 and 9.5%, respectively. It is not clear why total P fluxes are increasing (with corresponding smaller water fluxes), and the result suggests that the reduction target of 45%, relative to the 1980–1996 period, is close to 50% for the 2001–2005 period. Likewise, the 45% N load reduction target, relative to the 1980–1996 period, is equivalent to a 30% reduction relative to the 2001–2005 period. Fertilizer P consumption in the MARB has been relatively constant since about 1984 and is similar to consumption during 1970–1975. Net P inputs to the MARB have declined since the 1970s and have been predominantly negative since the mid-1990s (see Section 3.2 and Fig. 3.25). Table 4.1 also indicates N and P reduction recommendations in units of mass with reduction targets of 45% N and 45% P, assuming that the reduction was spread across all forms of N and P, that occur both annually and during the spring.

While the Study Group finds that both N and P reductions are warranted, additional modeling and dose–response research are needed to refine the reduction targets, particularly for P loading. Scavia and Donnelly (2007) presented the only model results that relate P loads to hypoxia in the NGOM. Further, there are no experimental data relating phytoplankton responses there to different levels of P. Ideally, targets for reducing P based on water quality should have greater model support and should consider dose–response relationships for P responses by the in situ phytoplankton communities. In the meantime, the response of the Gulf system to a specific amount of P reduction remains uncertain and must await the formulation of new models and dose–response relationships for the receiving waters. Water quality models aimed at evaluating the effects of these reductions will also rely on this information. Dose–response relationships should be developed using in situ bioassays designed to "ask the phytoplankton" what the response relationships and bloom thresholds are. These bioassay experiments are a logical follow-up to the work of Sylvan et al. (2006), which has shown P to be the limiting nutrient during periods of maximum primary production in the near-shore NGOM high-productivity zone. Bioassays are needed on a seasonal basis, where the effects of hydrologic variability and changing N:P input (loading) ratios on primary production, phytoplankton community composition, and biogeochemical and trophic fate can be evaluated.

In Section 4.5.8 on Most Effective Actions for Industrial and Municipal Sources, the Study Group provides some ballpark estimates of possible N and P reductions from upgrading major municipal wastewater treatment plants. The Study Group's example calculations demonstrate that sewage treatment plant upgrades to achieve total N concentration limits of 3 mg/L and total P concentrations of 0.3 mg/L could create reductions in total annual N flux to the Gulf by about 10% and the total spring N flux by about 6%. Upgrading to achieve P concentrations of 0.3 mg/L would create reductions in P fluxes from sewage treatment plants from 41,000 metric tons P/year (45,000 ton P/year) to 10,500 metric tons P/year (11,600 ton P/year) or about a 75% reduction in annual flux from sewage treatment plants to the MARB. These reductions, in turn, would translate into reductions of total annual P flux to the Gulf by about 20% and the total spring P flux by about 15%. If further investigation and data collection confirms the Study Group's calculations, upgrades to major wastewater treatment plants in the MARB could accomplish nearly half of the Study Group's recommended P reduction targets. This would represent very significant progress for both improving water quality in the MARB and reducing hypoxia in the NGOM.

Despite the need for additional model and bioassay work, the proposed target of a 45% reduction in annual P load should be used in an adaptive management framework to allow development of strategies that optimize both N and P reductions while more knowledge is acquired on P reduction impacts on near-field hypoxia. Unlike N, the P reduction strategy will help address water quality impairments in the MARB. Given the evidence that both N and P should be reduced in the NGOM, setting a goal for P reduction should not await the development of new models and availability of new experimental data. Enough information exists now to set a goal in an adaptive management context beginning with the P reductions that are already feasible given existing technologies and options.

In 2000, USEPA recommended nutrient criteria to states and tribes for use in establishing their water quality standards consistent with Section 303(c) of the Clean Water Act (CWA) (USEPA, 2000c). USEPA's recommended criteria represent an estimated "reference condition," and it is assumed that the reference condition concentration would protect all designated uses (including the most protected uses, such as high-quality fisheries, sensitive aquatic life). The Study Group asked USEPA for a comparison of the Study Group's recommended 45% reductions for TN and TP flux to the reductions in nutrient levels that would correspond to USEPA's ecoregional nutrient criteria for reference conditions (USEPA, 2006b). This comparison is provided in Appendix E. Although a number of assumptions were required to make this comparison (see the caveats in Appendix E), USEPA's preliminary analysis suggests that the Study Group's recommended targets for reducing TN and TP are, for most regions, not likely to be as stringent as would be obtained if states adopted USEPA's recommended reference condition values into state water quality standards for all waters. This comparison should not be interpreted as the Study Group's endorsement of USEPA's recommended nutrient criteria but rather an emphasis on the need to consider both within-basin nutrient criteria and NGOM load reduction goals. Numeric nutrient standards being developed by the states of the MARB will almost certainly be concentration rather than load based and may be most stringent during warmer, lower flow periods when absolute loads can be relatively low but when local waters are most frequently impaired by excess nutrient levels. It will be important for USEPA and other agencies to evaluate and, if necessary, reconcile within-basin water quality standards with load reduction goals for the NGOM. Strategies are needed for integrating standards throughout the MARB to better manage hypoxia as well as local water quality.

A mechanism in the Clean Water Act for addressing water quality impairments is the development of Total Maximum Daily Loads (TMDLs), though it is important to note that the focus of TMDL development is identification of the source and causes of water quality impairment, rather than on implementation of change for improving water quality. Under Section 303(d) of the Clean Water Act, states, territories, and authorized tribes are required to develop lists of impaired waters (i.e., waters that have not met water quality standards). The law requires that the appropriate jurisdictions develop TMDLs for these impaired waters. The TMDLs specify the maximum amounts of pollutants that waterbodies can receive and still meet water quality standards. In addition, TMDLs allocate pollutant loadings among point and nonpoint sources.

The status of nutrient criteria and TMDL development along the Mississippi River has been reviewed by the National Academy of Sciences (National Academy of Sciences, 2007). The National Academy of Sciences notes that none of the 10 Mississippi River mainstem states currently have numeric criteria for nitrogen or phosphorus applicable to the River and, that without such standards, there is little prospect of significantly reducing or eliminating hypoxia in the Gulf of Mexico. The National Academy of Sciences also describes how the process of developing numeric nutrient criteria and TMDLs for the Mississippi River could lead to water quality improvements in the Gulf of Mexico. NAS suggests that through such a process, USEPA could adopt the necessary numerical nutrient criteria for the terminus of the Mississippi River and waters of the northern Gulf of Mexico. Maximum nutrient loads could be assigned to each state and the loads could be translated into water quality criteria. Each state would then be required to develop a TMDL for waters that failed to meet the applicable criteria, and a coordinated effort could be undertaken to reduce point and nonpoint source loads to meet allocations established by the TMDLs. Thus, the NAS report identifies an approach through existing legislation (the Clean Water Act) that could be used to redress Gulf Hypoxia, but the SAB stresses that a great many steps exist between calling for "a coordinated effort" and implementing the full set of actions that must be undertaken for water quality to actually improve in the Gulf.

Key Findings and Recommendations

Based on findings since the *Integrated Assessment*, a N reduction target of greater than 30% will be needed to reduce the hypoxic area to 5,000 km^2 (1,930 mi^2). Recent research indicates N reductions of at least 45% will be needed to achieve the target in most years and reductions may have to exceed 50% due to effects of climate change. Research by several investigators provides evidence that P may limit primary production in the river outflow, near-field areas of the Gulf. Based on new research with the same model used to establish the N target, reductions in P loads of 40–50% are needed to reduce P-controlled hypoxia in the near-field areas of NGOM. P reductions in the MARB will not only benefit the NGOM but will also help to address P impairments in the MARB. Based on these findings, the Study Group offers the following recommendations.

- To reduce the size of the hypoxic zone, the total N flux to the NGOM from the combined Mississippi and Atchafalaya Rivers must be reduced by at least 45% from 1980 to 1996 average fluxes, to no more than 790,000 metric ton N/year (870,000 ton/year), and 290,000 metric ton N (320,000 ton) during the spring (April, May, June), both on a 5-year running average.
- To reduce the size of the hypoxic zone, commensurate reductions in P are needed. The total P flux to the NGOM from the combined Mississippi and Atchafalaya Rivers should be reduced by at least 45% from 1980 to 1996 average fluxes, to no more than 68,000 metric tonne P/year (75,000 ton P/year) on a 5-year running average.

4.3 Protecting Water Quality and Social Welfare in the Basin

The Study Group has been asked whether social welfare can be protected while reducing hypoxia and improving water quality in the Basin. To thoroughly answer this question would require quantification of the full costs of all activities undertaken to reduce the necessary nutrient loading into the Gulf (from agricultural sources, point sources, air deposition, etc.) and the full benefits accruing from those activities. The benefits would include the direct benefits of reducing the size of the hypoxic zone (commercial fishery effects, recreational fishery gains, the value placed on preserving intact ecosystems, biodiversity, etc.) and the "co-benefits" (such as improved local water quality, increased wildlife habitat, flood control, aesthetic values).

Since the costs, benefits, and co-benefits will depend on the extent of coverage and specific locations of control options, a complete answer to the question would require knowing the details of how such nutrient reductions would occur. For example, if these reductions are to be achieved entirely through restoration of wetlands and tighter municipal source controls, it would be necessary to know where the wetlands would be located and where the point-source reductions would occur in order to estimate their costs and their co-benefits. In contrast, an entirely different set of co-benefits and costs would likely result from relying on a broader array of control options that also included nutrient management, increased perennials, riparian buffers, drainage management, and reductions in air deposition. Further, the exact policy approach (e.g., expanded EQIP funding, mandates, or taxes) would need to be specified if estimates of the incidence of the costs are to be estimated (i.e., whether the costs would ultimately be borne by taxpayers, by consumers, or by farmers and landowners).

To date, no set of models and/or studies have been undertaken that address all of the necessary components on a basin-wide scale to estimate the effects on social welfare. However, a number of studies, beginning with the research in the *Integrated Assessment*, have been done that address substantial components of this question. More complete efforts at quantifying the control costs than the benefits have been undertaken, though there remains a need for much more work on both sides of the equation. Integrated models at multiple levels and scales are needed to support this effort. The existing research focuses largely on agricultural nonpoint source control. This section summarizes findings from the limited set of large-scale economic-watershed models of agricultural nonpoint sources that have been applied to date.

4.3.1 Assessment and Review of the Cost Estimates from the CENR Integrated Assessment

Doering et al. (1999) in the *Integrated Assessment* undertook an ambitious cost-effectiveness analysis of several policy approaches to reach the N loss reduction goal of 20% established as part of the *Integrated Assessment*. The central modeling system they used was the US Mathematical Programming (USMP) model, which represents the agricultural sector in 45 production regions throughout the United States with 10 crops, 16 animal products, retail and processed products, and a range of domestic and international supply and demand relationships. Management practices include crop rotations, five tillage options, and varying fertilizer rates.

The environmental effects of various management practices and land uses in USMP are predicted by the EPIC model (the Environment Productivity Impact Calculator). USMP uses EPIC to predict changes in N loss, P loss, and sediment loss at the edge of the field from changes in land use and conservation practices. Donner et al. (2002) chose a 20% N loss reduction goal as "the best combination of sizable nitrogen loss reductions and acceptable economic costs" (Doering et al., 1999 p. 37). The remainder of their analyses focused on the evaluation of several policies that might achieve this environmental goal. Some key predictions from the modeling system include

- A 20% reduction in fertilizer N application rates would result in the reduction of edge-of-field N loss by about 11%. In contrast, a 45% reduction mandate and fertilizer tax set to achieve a 45% reduction is predicted to result in the target goal of N loss reduction of about 20%. The less than proportional reduction in N loss coming from reduced fertilization in this modeling system is a result of predicted changes in acreage resulting from the feedback effect of price changes. Specifically, higher crop prices due to lower yields from the reduced fertilization rates induce more acreage planted to the fertilized crop, thereby partially offsetting the reduction in N. Whether the magnitude of the yield effects embedded in these models is accurate is an important question. For further discussion of this issue, see Section 4.5.6.

- Some 7.29 million hectares (18 million acres) of wetland restoration would achieve the 20% reduction in N loss goal at a cost of over $30 billion.
- Restoration of 10.9 million hectares (27 million acres) of riparian buffers was estimated to cost over $40 billion and generated relatively small reductions in N losses, suggesting that this strategy is not cost-effective for hypoxic zone control. In light of current evidence that phosphorous is also of concern, this result should be reconsidered as there is significant evidence that buffers can be quite effective in reducing sediment and phosphorous loss.
- A "mixed policy" with a 2.02 million hectares (5 million acres) wetland restoration program in conjunction with a 20% fertilizer reduction is more cost-effective than most of the previous approaches, but the 45% reduction in fertilizer is more cost-effective yet.
- The introduction of point–nonpoint source trading across the basin where the cap applies only to point sources will not achieve the 20% N loss reduction due to the relatively small magnitude of N contribution from point sources. Even with a stringent standard on point sources, only about 5% of the needed reductions occur.
- These policies are likely to produce large "co-benefits" (i.e., other environmental benefits occurring within the basin and on-farm productivity benefits not immediately captured in the current profitability resulting from the policies). For example, the authors estimate that restoration of 405,000 hectares (1,000,000 acres) of wetlands would yield total benefits in the basin that exceed the costs, even without considering any benefits of hypoxia reduction.

Cost estimates used for the *Integrated Assessment* for a 20% reduction in N discharge coming from agricultural nonpoint sources range from $15 billion to $30 billion; however, these estimates suffer from a number of shortcomings including consideration of only a few options for reducing nutrient discharge and limited targeting. More inclusive assessments with better targeting of options to locations where they are most appropriate may reduce these costs.

In follow-up research, some of the same study coauthors (Ribaudo et al., 2001) compare nitrogen reduction methods with wetland restoration and low and high levels of N loss reduction. They find that nutrient management is more cost-effective at low levels of N loss reduction while wetlands restoration is more cost-effective at high levels. Tables 4.2 and 4.3 (listed at the end of this discussion) briefly summarize the key components of these studies and the other large-scale studies that are reviewed in the following discussion.

Due to limits on the understanding of the economics and natural science at the time, the work in the *Integrated Assessment* and its follow-up is based on assumptions that, in light of more recent research and availability of data, assessments could be improved upon in future work. The USMP model represents a wide variety of agricultural raw inputs and intermediate products at a relatively aggregate scale. However, it does not contain detailed description of land use, soil characteristics,

Table 4.2 Summary of study features of basin-wide integrated economic-biophysical models

Authors	Study region	Models used	Environmental measures	Comments
Doering et al. (1999)	Entire United States with policies simulated in Mississippi River basin	USMP and EPIC	N, P, and sediment in the MARB (not delivered to the NGOM)	Original CENR study
Ribaudo et al. (2001)	Entire United States with policies simulated in Mississippi River basin	USMP and EPIC	N, P, and sediment in the MARB (not delivered to the NGOM)	Extension of CENR study
Greenhalgh and Sauer (2003)	Entire United States with policies simulated in Mississippi River basin	USMP and EPIC with SPARROW derived transport coefficients	N delivered to the Gulf, greenhouse gas emissions, P and N, soil erosion in the MARB	Study focuses on co-benefits of policies
Wu et al. (2004)	Upper Mississippi River basin	Econometric model and EPIC based metamodels	N leaching, N runoff, wind erosion, and water erosion in UMRB	Finer spatial detail than USMP but no price feedbacks
Ribaudo et al. (2005)	Entire United States with policies simulated in Mississippi River basin	USMP and EPIC	N in MARB	Follow-up to original CENR study
Wu and Tanaka (2005)	Upper Mississippi River basin	Econometric model and SWAT	N delivered to the NGOM	Finer spatial detail than USMP but no price feedbacks
Kling et al. (2006)	Upper Mississippi River basin	Econometric model and SWAT	N, P, and sediment in UMRB and N delivery to the NGOM	Finer spatial, but no price feedbacks

Table 4.3 Summary of policies and findings from integrated economic-biophysical models

Study	Policies/actions evaluated	Key findings[a]
Doering et al. (1999)	1. Fertilizer reduction mandates/fertilizer taxes 2. Wetland restoration 3. Riparian buffer 4. Mixed policy (wetlands and fertilizer reduction) 5. Water quality trading	1. Cost-effective approaches exist to reducing nitrogen losses in the 20% range 2. Wetland-based strategies are more expensive than fertilizer reduction 3. Buffers are not cost-effective for reducing N losses 4. A combination of 5 million acres wetland restoration with 20% fertilizer reduction is most cost-effective 5. These cost-effectiveness measures do not take into account the transport of nitrogen to the Gulf and the rankings of preferred alternatives could change
Ribaudo et al. (2001)	1. Reduce fertilizer rates 2. Wetland restoration	1. Below 26% reduction in N losses, fertilizer reduction/management is most cost-effective 2. Above this rate, wetland restoration is most cost-effective
Greenhalgh and Sauer (2003)	1. N trading between point and nonpoint sources 2. Greenhouse gas trading 3. N trading with additional payments for GHG reduction 4. N fertilizer tax 5. Conservation tillage payment 6. Expansion of CRP to 40 million acres nationwide	1. Nutrient trading (point/nonpoint) with tighter discharge limits could reduce nitrogen reach the NGOM by 11% annually 2. Nutrient and greenhouse gas trading were the lowest cost policies, but nutrient trading was the most cost-effective 3. The co-benefits of these policies in terms of greenhouse gas reductions, phosphorous, and sediment can be significant
Wu et al. (2004)	1. Conservation payments for conservation tillage 2. Crop rotations	Crop rotations not a cost-effective strategy for N reduction
Wu and Tanaka (2005)	1. Fertilizer tax 2. Payments for conservation tillage 3. Payments for land retirement 4. Payments for crop rotations	Fertilizer tax is the most cost-effective of policies considered
Booth and Campbell (2007)	Targeting CRP to watersheds with the greater proportion of fertilizer used. Hence, CRP rises in direct proportion to fertilizer/cropping intensity	Targeting CRP and enrolling an additional 2.7 million hectares in those areas with the greatest fertilizer intensity would increase annual agricultural subsidies to the MARB by 6.2% (over the combined commodity support and conservation funding in 2003)

Table 4.3 (continued)

Study	Policies/actions evaluated	Key findings[a]
Ribaudo et al. (2005)	N trading between point and nonpoint sources	Trading between waste water treatment plants and nonpoint/agricultural sources to meet the reductions achievable by installing advance nutrient removal technology at treatment plants would have large welfare gains
Kling et al. (2006)	Implementation of a set of targeted conservation practices including conservation tillage, land retirement, terraces, contouring, grassed waterways, and reduce fertilization rate on corn	1. Annual costs of \$800 million per is predicted to achieve 22% reduction in N loading to the NGOM 2. Within the UMRB, sediment loads were reduced by 40–66%, total P was reduced by 6–47%, and N by 9–29%

[a]Doering et al. (1999) also conclude that fertilizer restrictions are more cost-effective than a fertilizer tax, but they apparently incorrectly count tax revenues as a cost rather than a transfer. The restrictions and tax have the same welfare effects, though different distributional implications.

yields, etc., at the individual field and/or subbasin scale. This inability to target finer scales could result in overstating the costs of meeting a particular reduction goal because significant cost savings can accrue from targeting land-management strategies.

The *Integrated Assessment* assumed a one-to-one relationship between the reduction in edge-of-field nitrogen loss and reduced loadings to waterways without incorporating the geographic differences in movement of N from the field of origination to the Gulf. Whether this shortcoming over- or understates the costs is an empirical question, but the results coming from a model that explicitly incorporates the fate and transport of nutrients and sediment might suggest very different results concerning the cost-effectiveness.

4.3.2 Other Large-Scale Integrated Economic and Biophysical Models for Agricultural Nonpoint Sources

Since completion of the *Integrated Assessment*, several basin-wide studies have evaluated policies that might reduce Gulf hypoxia and/or have effects on other environmental amenities that could be considered co-benefits (including carbon sequestration and upstream, local water quality indicators). The models can be divided into those that use the USMP modeling framework and those based on econometric estimates of behavioral response to economic drivers.

Booth and Campbell (2007) used a regression model to estimate the cost of reducing N losses when targeting conservation dollars to those areas with the highest proportion of fertilizer use. They modeled a hypothetical case in which conservation enrollment rises in direct proportion to the nonlinear rise in nitrate flux that occurs as fertilization intensity increases. The result was an increase in the amount of land in the high fertilizer watersheds enrolled in the Conservation Reserve Program by 2.7 million hectares (6.67 million acres) (a 29% increase over 2003 CRP levels) at a cost of $448 million. Booth and Campbell (2007) describe this as a 6.2% increase over the combined cost of commodity support and conservation programs. They account for the drop in commodity support spending that would accompany the enrollment of commodity-farmed land in the CRP. Booth and Campbell (2007) do not specify the percentage reduction in nitrate loading that would result from this scenario.

Wu et al. (2004) and Wu and Tanaka (2005) developed an econometric model of crop choice and tillage choice using the National Resources Inventory for the upper Mississippi River basin. They estimated the probability of adopting conservation tillage and crop choice based on a variety of physical and economic variables including land quality, slope, climate conditions, and profits. They used over 40,000 crop land points observed for 16 years, although only a subset of the observations were used for model fitting. These adoption models then simulate adoption profiles under alternative policies. Finally, the environmental effects of the policies are predicted with a biophysical model. Wu et al. (2004) used a set of environmental production functions estimated via a meta-modeling approach (Wu and Babcock, 1999), based on data generated from the EPIC model. They found that crop rotations are not a cost-effective strategy to N reduction.

Wu and Tanaka (2005) used the SWAT model to predict water quality changes from the policies. They considered the same two policies as Wu et al. (2004), as well as a policy that would increase the amount of land set-aside in a Conservation Reserve-type program and a fertilizer tax at various rates. They found a fertilizer tax to be the most cost-effective of policies they considered.

Kling et al. (2006) employed a similar econometric modeling approach. Like Wu et al. (2004), they used the National Resource Inventory data to link the cost data with the SWAT model. They estimated the costs and water quality benefits of implementing a set of conservation practices associated with implementation rules based on distances to a waterway, slope, and erodibility indices. The conservation practices assessed include grassed waterways, nitrogen management, terraces, buffers, land retirement, and conservation tillage. They estimated that this placement of conservation practices on the landscape would cost over $800 million annually (or roughly $16 billion if viewed as a lump sum cost assuming a 5% rate of discount) and would achieve a 22% reduction in N loadings into the upper Mississippi River basin at Grafton, IL. Within the UMRB, they estimated a 40–66% reduction in sediment loads, a 6–47% reduction in P loads, and a 9–29% reduction in N loads. These estimates (like those from all of the studies reviewed here) are likely to be very sensitive to the set of conservation practices included and the specific scenarios studied.

Greenhalgh and Sauer (2003) used the USMP augmented in two important ways: (1) they configured the model by watersheds and added information on municipal waste water treatment plants and (2) they included "attenuation" coefficients derived from the SPARROW model to reflect the transport component of N flows between watersheds. The focus of their work was on policy options for hypoxia that also contribute to greenhouse gas reductions. The policies they considered include N trading between point and nonpoint sources, GHG trading assuming external carbon prices of \$5/ton and \$14/ton, N trading with additional payments for GHG emission reductions, an N fertilizer tax, a subsidy to farmers willing to shift from conventional to conservation tillage, and an expansion of the CRP program to 16 million hectares (40 million acres) nationwide. Of the policies evaluated, none achieved the 20% reduction goal of the Doering et al. (1999) analysis. The largest reductions were achieved in their simulation of point/nonpoint source trading with a stringent N standard. The most cost-effective policies were also the trading programs.

Ribaudo et al. (2005) also considered the possibility of N trading between point and nonpoint sources using the USMP model. They found that trading has significant potential to reduce costs relative to a requirement that wastewater treatment plants be required to install stringent nutrient removal technology. [1]

These studies shed light on the costs of addressing the hypoxia problem from conservation practices in the agricultural sector and the way these costs may vary depending on the policy instrument chosen (trading program, conservation payment, tax, etc.). These studies also directly bear on the question of how much it will cost to address local water quality in the MARB. However, as noted above, shortcomings of the integrated models have prevented assessment of many policies as well as conservation practices and sinks. None of the models include point source and nonpoint source control options. With the exception of Booth and Campbell (2007), most models have not adequately addressed the cost savings associated with targeting. Nonetheless, results to date suggest that there is large variability in the costs of alternative policies. The issue of who pays these costs may also be important to consider since the incidence (who must pay the costs) may differ dramatically across policies. A notable example is a fertilizer tax, which has the same social costs as a restriction but which may have a much higher incidence on farmers.

Improved estimates of the costs of installing and maintaining conservation practices could be generated with the current suite of models by considering alternative sets of conservation practices. This can be accomplished using the following steps: (1) identifying conservation practices that are most likely to be effective in reducing nutrients important for hypoxia and (2) identifying scenarios that place these conservation practices on the landscape. These scenarios could be based on rules of thumb (identifying, for example, a particular conservation practice to be used on cropland with specific climate and soil characteristics); algorithms for optimal placement to minimize costs; multiple goals, such as maximizing in basin co-benefits or income support; or policy-relevant methods, such as the use of an environmental benefits index; or computing cost estimates from economic models and water quality changes from watershed models.

[1] It is important to recognize that these studies assume a perfectly efficient water quality trading program with no trading restrictions; current water quality trading programs do not match the modeled system

4.3.3 Research Assessing the Basin-Wide Co-benefits

As noted above, many of the same practices that could contribute to reductions in the hypoxic zone could also have significant effects on local water quality, carbon sequestration, wildlife habitat, flood control, and other ecosystem services. The physical co-benefits of many conservation practices and sinks are described in Section 4.5.10. On the basin-wide scale, there are a few studies that provide physical measures of one or more co-benefits that are associated with implementation of conservation practices that would address hypoxia, particularly related to carbon sequestration and water quality (see, for example, Feng et al. (2005), Lewandrowski et al. (2004), Greenhalgh and Sauer (2003)). These studies consistently indicated that significant co-benefits are present, but these estimates are not monetized and are reported in physical units. Further, the policies analyzed are not focused on hypoxia reduction.

Thus, the work reported in the *Integrated Assessment* remains the most complete coverage to date of the potential value to MARB residents of the water quality and other co-benefits. The estimates provided there suggested that the monetized value of the benefits to the basin were larger than the costs based primarily on benefit estimates of the value of erosion control and wetlands restoration. A more complete accounting of these benefits could be developed using benefits transfer techniques, although there are many ecosystem services for which currently accepted methods are not likely to adequately fully capture the value of the benefits. But, in any case, because the *Integrated Assessment* was not able to quantify all co-benefits, total co-benefits within the basin would almost certainly be larger than those estimated.

Due to the incredible complexity in this system, as well as limits in data, modeling, and research, definitive statements on social welfare are not possible. For example, there is incomplete information on the costs of farm-level actions to reduce edge-of-field nutrient losses. There is even greater uncertainty in quantifying the effectiveness of farm-level nutrient control actions in reducing watershed-level nutrient flux and about the relationship between watershed-level nutrient flux and the spatial and temporal dimensions of the hypoxic zone. These uncertainties are further exacerbated by the possibility of regime shift in the Gulf of Mexico, whereby the system could become more susceptible to hypoxia following the initial occurrences. If regime shift is a factor, then historic data on the relationship between nutrient flux and the size of the hypoxic zone does not provide guidance on the decrease in nutrients required to achieve a given reduction in the size of the hypoxic zone. Hence, a return to historic lower levels of nutrient fluxes might not be adequate to return to a corresponding size of the hypoxic zone.

There are many sources of uncertainty in the economic, hydrologic, and Gulf systems that make it difficult to render definitive conclusions about social welfare. Indeed, it is precisely because of these many uncertainties and need for additional research that we recommend an approach based on an adaptive management strategy that aims to move in a "directionally correct" fashion, rather focusing on achieving a precise outcome.

While we cannot definitely say that we can achieve the 5,000 km^2 (1,930 mi^2) goal while maintaining social welfare, there is evidence that suggests it is feasible to do so. First, and perhaps most importantly, welfare losses in the Basin will be at least partially or even totally offset by co-benefits of nutrient reduction actions. For example, if wetlands restoration is used to control nutrient flux, it will result in improvements in wildlife habitat and local water quality, both of which will improve welfare in the Basin. Findings from the Doering et al. (1999) assessment point out that the benefits accruing locally from wetlands restoration might well exceed the costs, even without any Gulf hypoxia reductions. Similar estimates are reported in Hey et al. (2004) for substantial restoration of wetlands in flood plains (see Section 4.4.2). Management actions that reduce farm-level nutrient losses may lead to better local water quality, thereby improving welfare for affected residents within the Basin. If management actions are undertaken to control air emissions, thereby reducing atmospheric deposition of nitrogen, it will result in improvements in air quality, reduction in acid precipitation, lower emissions of greenhouse gasses, etc. Thus, co-benefits within the Basin will at least partially and perhaps fully offset welfare losses associated with the costs of implementing management actions. And in the longer term, a transition from corn to perennial crops could benefit farmers and other Basin residents. Thus, there may be larger scale transitions in the agronomic system that provides opportunities to reduce nutrient flux while maintaining welfare in the Basin.

A second reason for optimism is that cost-effective approaches, such as targeting low cost sources and using emissions trading, have not yet been applied. These approaches have the potential to reduce the costs of nutrient control, possibly considerably, thereby reducing the burden of complying with the goal. Thus, there may be opportunities to control the cost of nutrient reduction.

4.3.4 Principles of Landscape Design

Another perspective for protecting social welfare can be drawn from the principles of landscape design. A landscape perspective involves broad-scale consideration of how decisions affect resources, particularly in the long run. Guidelines have been proposed as a way to facilitate land managers considering the ecological ramifications of land-use decisions (Dale et al., 2000). These guidelines are meant to be flexible and to apply to diverse land-use situations, yet require that decisions be made within an appropriate spatial and temporal context. These landscape design guidelines can serve as a checklist of factors to be considered in making decisions that relate to implications for hypoxia in the Gulf.

- *Examine the impacts of local decisions in a regional context.* The spatial array of habitats and ecosystems shapes local conditions and responses (e.g., Patterson, 1987; Risser, 1985) and local changes can have broad-scale impacts over the landscape. Hypoxia is a classic example of such impacts (Russo et al., 2008), for fertilizer applications in the Midwestern states can affect oxygen conditions in the Gulf of Mexico. This guideline notes that it is critical to examine both the constraints placed on a location by the regional conditions and the implications of decisions for the larger area. Therefore, it is critical to identify the surrounding region that is likely to affect and be affected by the decision and examine how adjoining jurisdictions are using and managing their lands. Forman (1995) suggests that land-use planning should first determine nature's arrangement of landscape elements and land cover and then consider optimal spatial arrangements and existing human uses. Following this initial step, he suggests that the desired landscape mosaic be planned first for water and biodiversity; then for cultivation, grazing, and wood products; then for sewage and other wastes; and finally for homes and industry. Of course, planning under pristine conditions is typically not possible. Rather, the extant state of development of the region generally constrains opportunities for land management.

- *Plan for long-term change and unexpected events*. Impacts of decisions can, and often do, vary over time as a result of delayed and cumulative effects. Future options are often constrained by the decisions made today as well as by those made in the past. For example, areas that are urbanized are unlikely to be available for any other land uses because urbanization locks in a pattern on the landscape that is hard to reverse. Thus, management actions should be implemented with some consideration as to the physical, biological, esthetic, or economic constraints that are placed on future uses of resources. External effects can extend beyond the boundaries of individual ownership and thus have the potential to affect surrounding owners. Planning for the long term also requires consideration of the potential for unexpected events, such as variations in temperature or precipitation patterns or disturbances. Long-term planning must also recognize that one cannot simply extrapolate historical land-use impacts forward to predict future consequences of land use. The transitions of land from one use or cover type to another often are not stable over time because of changes in demographics, public policy, market economies, and technological and ecological factors.
- *Preserve rare landscape elements, critical habitats, and associated species.* This guideline implies a hierarchy of flexibility, and it implicitly recognizes ecological constraints as the primary determinants in this hierarchy. For example, a viable housing site is much more flexible in placement than an agricultural area or a wetland dedicated to improving water quality and sustaining wildlife. Optimizing concurrently for several objectives requires that planners recognize lower site flexibility of some uses than others. However, given that most situations involve existing land uses and built structures, this guideline calls for examining local decisions within the regional context of ecological concerns as well as in relation to the social, economic, and political perspectives that are typically considered.
- *Avoid land uses that deplete natural resources over a broad area.* Depletion of natural resources disrupts natural processes in ways that often are irreversible over long periods of time. The loss of soil via erosion that can occur during agriculture and the loss of wetlands and their associated ecological processes and species are two examples. This guideline requires the determination of resources at risk, which is an ongoing process as the abundance and distribution of resources change. This guideline also calls for the deliberation of ways to

avoid actions that would jeopardize natural resources and recognition that some land actions are inappropriate in a particular setting or time, and they should be avoided.

- *Avoid or compensate for effects of land use on ecological processes.* Negative impacts of land-use practices might be avoided or mitigated by some forethought. To do so, potential impacts need to be examined at the appropriate scale. At a fine scale, farm practices may interrupt ecoregional processes. At a broad scale, patterns of watershed processes may be altered, for example, by changing drainage patterns as part of the land use. Therefore, how proposed actions might affect other systems (or lands) should be examined. For example, human uses of the land should avoid uses that might have a negative impact on other systems; at the very least, ways to compensate for those anticipated effects should be determined. It is useful to look for opportunities to design land use to benefit or enhance the ecological attributes of a region.
- *Implement land-use and -management practices that are compatible with the natural potential of the area*. Local physical and biotic conditions affect ecological processes. Therefore, the natural potential for productivity and for nutrient and water cycling partially determine the appropriate land-use and management practices for a site. Land-use practices that fall within these limits are usually cost-effective in terms of human resources and future costs caused by unwarranted changes on the land. Nevertheless, supplementing the natural resources of an area by adding nutrients through fertilization or water via irrigation is common. Even with such supplements, however, cost-effective management recognizes natural limitations of a site. Implementing land-use and -management practices that are compatible with the natural potential of the area requires that land managers understand a site's potential. For example, land-management practices such as no-till farming reduce soil erosion or mitigate other resource losses. Often, however, land uses ignore site limitations or externalize site potential. For example, building shopping malls on prime agriculture land does not make the best use of the site potential. Nevertheless, land products are limited by the natural potential of the site.

Together these guidelines form the basis of a landscape design perspective that should improve the ability to understand and manage the complex system that is affecting hypoxia in the Gulf of Mexico.

Key Findings and Recommendations

The large-scale policy models that have been developed to date each have strengths and weaknesses. None of the models adequately address the full range of management options (wetlands, buffers, nutrient management, etc.) or the full range of policy instruments in a geographically explicit manner. In fact, no single model is likely to be adequate for the full range of decision making that adaptive management of this complex system requires. Moreover, the focus of prior analyses was on cost-effective strategies to reduce N loss, which was the concern at the time. Given that the best current science suggests P is also a limiting nutrient in the Gulf, it is important to seek cost-effective practices that affect both N and P while considering possible trade-offs between them.

The CENR study remains the only research effort to consider the overall costs and benefits of controlling hypoxia in the Gulf of Mexico. The study suffers from a number of shortcomings (many control options and sources of nutrients were not considered, the hydrology of fate and transport was ignored, and no sensitivity analysis concerning key assumptions was undertaken to name a few). The evidence from this work and other studies suggests that it is probable that social welfare in the basin can be maintained while achieving the goal of a 5-year running average of 5,000 km^2 for the hypoxic zone. Most importantly, welfare losses from costs incurred to control hypoxia in the Basin will be offset, at least in part, by co-benefits of nutrient reductions. For example, research on wetlands in the MARB suggests that the benefits of large-scale restoration efforts would exceed the costs. Second, only limited targeting of control options that focus on hypoxia reduction and its co-benefits have been undertaken. Given the significant gains in cost savings that targeting can achieve, this suggests that it may be possible to achieve hypoxia reduction at lower cost than predicted in models that do not consider complete targeting. Based on these findings, the Study Group offers the following recommendations.

- The management of factors affecting hypoxia within the MARB should be viewed as components of a designed landscape so that costs and benefits at various spatial and temporal scales are explicitly considered.
- Integrated economic and watershed models are needed to support an adaptive management framework. Models are needed that represent land use and costs of conservation at both the fine scale, such as the 8 or 12-digit HUC size, as well as a larger scale that encompasses the entire MARB.
- Research that assesses the optimal suites of conservation practices to maximize both local water quality and other co-benefits and Gulf hypoxia reduction is needed. This will require improved understanding of the watershed-scale benefits of these control measures and their costs.
- To reduce hypoxia and protect social welfare in the MARB, control measures that both reduce hypoxia cost-effectively and provide co-benefits in the MARB should be targeted whenever possible. Targeting control measures can reduce the costs and increase co-benefits associated with measures to control hypoxia in the Gulf of Mexico.

4.4 Cost-Effective Approaches for Nonpoint Source Control

While the *Action Plan* and this Study Group urge the reliance on adaptive management principles, a variety of tools can be used as the vehicle for implementation within adaptive management. The current *Action Plan* indicates a principle of encouraging "actions that are voluntary, practical, and cost-effective" (page 9). Additionally, the plan will "utilize existing programs, including existing State and Federal regulatory mechanisms," as well as identify needs for additional funding. These statements include a variety of tools ranging from purely voluntary programs (those with no associated financial incentives) to current conservation programs funded by state and federal agencies (such as the Conservation Reserve Program [CRP] and the Environmental Quality Incentive Program [EQIP]) to water quality trading. Research assessing the costs and effectiveness of these approaches is addressed in this section.

Complicating the design of cost-effective approaches is the geographic distance between the sources of nutrients and the receiving waters downstream. Two identical farm fields in different locations (with resulting differences in the hydrology of the local watershed) will send differing amounts of nutrients to the Gulf. Hence, the effectiveness of a practice or sink in a particular location depends on what sources and sinks are present elsewhere in the watershed. Whether it is cost-effective to install a buffer at a particular location may depend upon whether there is a wetland at the base of the watershed, whether conservation tillage is being practiced elsewhere, etc. Thus, rather than focus on individual practices, policy options that can simultaneously encourage the adoption of practices and sinks that are jointly cost-effective will best protect social welfare in the Basin.

It is important to clarify the concept of "costs." Here, "costs" refers to the least amount of compensation needed to effect change, e.g., the compensation that would be necessary for a landowner or farmer to adopt a conservation practice. This is the standard concept of economic cost, relevant to any good or service. This cost includes "direct" costs, such as the cost of new equipment, building of structures, and labor to manage a practice, as well as a myriad of potential "indirect" costs, such as lost profits from adopting the practice and compensation for added risk from the practice. Components of these costs can be negative; that is, it may actually increase profitability to adopt some practices (conservation tillage in certain circumstances is a notable example).

Second, the focus of most economic studies is on total costs with little or no consideration paid to what subset of society actually bears the costs (incidence) of the policy. This focus on efficiency (seeking the lowest cost approach) is based on the premise that compensation could always be paid to those bearing the cost in some form so that society will be best off if the lowest cost option is pursued. However, since such compensations are rarely paid, the issue of who pays is likely to enter the policy decision. Complete information on the incidence of alternative tools in this context is not available, but where appropriate, we note the likely incidence considerations.

4.4.1 Voluntary Programs – Without Economic Incentives

There is a small and growing literature concerning the effectiveness and optimal design of voluntary agreements that do not have positive or negative financial incentives associated with them (Morgenstern and Pizer, 2007; National Research Council, 2002). Key insights were presented in a game-theoretic model by Segerson and Miceli (1998), who identified the conditions under which voluntary agreements are likely to yield efficient pollution levels without significant economic incentives. They studied voluntary agreements that are based on threats of harsher outcomes if the goals are not met, using the example of mandatory abatement requirements if the voluntary agreement does not succeed in meeting the pollution goal. The premise is that firms will voluntarily agree to reduce pollution if they can avoid the costs that future mandatory controls would otherwise bring. In the absence of financial compensation, the presence of a positive probability of a penalty (or cost in the form of mandatory control) is required to support Segerson and Miceli's findings that there are situations in which efficient levels of pollution control can be achieved with voluntary agreements (without economic incentives). They found that pollution reduction is likely to be small when the background threat is weak.

Empirical work also sheds light on the efficacy of voluntary agreements that do not have financial incentives. Mazurek (2002) identified 42 voluntary environmental initiatives sponsored by the federal government since 1988. Although the programs she identified are largely outside the realm of agriculture, her conclusions are relevant. Mazurek concluded that a variety of implementation problems have led to "lower-than-expected" environmental results for voluntary (without financial incentive) agreements, a result consistent with findings of a 1997 USGAO (1997) report concerning four voluntary agreements related to climate change.

In the same National Research Council report (2002), Randall identified three essential functions for government if voluntary agreements (without financial incentives) are to be effective. These key functions are meaningful monitoring to back up a threat of government inspection, "credible threat of regulation" if the goals are not met, and a clear liability system to punish "blatant polluters and repeat offenders." Randall concluded that "voluntary (or negotiated) agreements, industry codes, and green marketing should be viewed as promising additions to the environmental toolkit, but they should supplement, not supplant, the regulatory framework. They make a nice frosting on the regulatory cake. But the cake itself must be there."

Finally, Morgenstern and Pizer (2007) presented seven case studies on voluntary agreements (without economic incentives) in the United States and elsewhere. Point estimates of environmental improvements attributable to the voluntary programs ranged from negative values (actual declines in environmental performance) to a maximum of 28% improvement in environmental performance. Morgenstern and Pizer concluded "that voluntary programs have a real but limited quantitative effect. . .."

Given the historical aversion to imposing mandatory requirements in agriculture, the collective weight of these studies suggest that voluntary agreements that do not have incentives associated with them are not likely to be adequate on their own to achieve significant reductions in nutrient runoff. In short, voluntary programs without incentives can have small effects but cannot be relied upon to induce major environmental improvements.

4.4.2 Existing Agricultural Conservation Programs

Currently, the largest incentive-based conservation programs related to agriculture are the EQIP and CRP. A potentially significant program introduced in the 2002 Farm Bill was the Conservation Security Program (CSP), which has been funded only partially and implemented incrementally. The CRP pays farmers to retire land, and the other two pay farmers to implement conservation practices on their farms (EQIP is a cost-share program; CSP was intended to cover the full costs of adoption). Numerous studies undertaken by USDA's Economic Research Service and others have estimated the magnitude of environmental benefits from these programs in physical terms (e.g., tons of erosion reduction, acres of habitat preserved, acres of wetlands restored) and some efforts have been made to monetize these benefits [see Claassen et al. (2004) for a summary of CRP studies as well as Haufler (2005)]. The Conservation Effects Assessment Program (CEAP) was initiated in an attempt to provide nationwide estimates of the benefits provided by the full suite of conservation programs; a national assessment of the water quality benefits is being developed currently (Bob Kellogg, presentation to SAB Hypoxia Advisory Panel, December 6, 2006).

The CRP pays landowners to take their land out of crop production and place it in perennial vegetation or trees, depending on the region of the country, with a goal of creating wildlife habitat and reducing erosion (and originally to reduce crop production). The CRP enrolls about 10% of total US cropland, nearly all in 10-year contracts although there is significant concern that high corn prices due to ethanol expansion may rapidly reduce this amount. A number of studies have identified large environmental benefits associated with the CRP [Smith and Alexander (2000), Feather et al. (1999)]. The program has used an Environmental Benefits Index (EBI) since 1990 to prioritize parcels for inclusion in the program that gives points to land based on particular environmental attributes and cost. The movement from targeting erodible lands (prior to 1990) to the use of the EBI for targeting has been estimated to have doubled the benefits from the program (Feather et al., 1999). Ribaudo (1989) estimated that a CRP enrollment that targets lands based on environmental damages (benefits) would have significantly greater benefits still. By redesigning the weights in this index, the program could target land that is predicted to contribute high nutrient loadings to the Gulf.

Many other studies have addressed the cost-effectiveness of land retirement to achieve environmental benefits within the context of the CRP. In a series of papers assessing the efficiency of the Conservation Reserve Enhancement Program (CREP) in Illinois, Khanna et al. (2003) linked the AGNPS model with site-specific characteristics of parcels to examine the relative efficiency of alternative targeting mechanisms (Yang et al., 2003, 2004, 2005). Extremely large gains from targeting were reported; for example, Yang et al. (2004) estimated that with targeting, 30% less cropland could have been retired (at almost 40% less total cost), while achieving 20% reductions in erosion instead of the actual 12% reduction.

The EQIP program is a cost-share program for conservation practices in livestock facilities and on land that remains in agricultural production. A prospective benefit cost analysis (as required by Executive Order 12866) predicted over $5 billion in net benefits from the EQIP program as implemented under the 2002 Farm Bill, even though not all of the benefits could be monetized (US Department of Agriculture, 2003).

The Wetland Reserve Program (WRP), Grassland Reserve Program (GRP), and Wildlife Habitat Incentive Program (WHIP) are all smaller land retirement programs that also could potentially benefit efforts to reduce Gulf hypoxia. Additional information on the large-scale potential for wetlands is provided by Hey et al. (2004), who addressed the question of whether the social benefits from restoring up to 2.83 million hectares (7 million acres) of cropland in the 100 years floodplain of the upper Mississippi River basin to wetlands exceed the costs. The benefits include reduced flood-related crop damages; reduced crop subsidies; and non-flood-related recreation benefits of wetland conversion, including fishing, hunting, and general recreation usage. These benefits were compared to estimates of the costs of cropland conversion comprised of farm rental rates (representing the present value of farmland income) and the costs of wetland construction and maintenance. Hey et al. (2004) estimated that the benefits exceed the costs in all locations considered except one county in Missouri. In the context of NGOM hypoxia, this difference is especially striking because the benefits exceed the costs for this conversion even without considering any benefits from reduction of the hypoxic zone. As the authors carefully pointed out, the social efficiency of converting 2.83 million hectares (7 million acres) does not mean that private benefits will exceed the private costs for all parties. Individual landowners would stand to lose while recreationists accrue benefits.

These findings represent an important addition to the assessment of wetlands in the *Integrated Assessment*. While Doering et al. (1999) concluded that wetland restoration was less cost-effective than fertilizer reductions, their analysis did not include cost savings from crop subsidy reductions nor flood-related crop damages. In addition, the Hey et al. (2004) work focused on wetlands targeted in flood plains. The study suggests two points of key importance for NGOM hypoxia: (1) there is a large amount of acreage that is situated in locations that potentially could serve as nutrient sinks in the upper Mississippi River basin, and (2) the co-benefits of this action are large enough, in and of themselves, to justify the social efficiency of converting this land to nutrient sinks even without considering the benefits associated with reducing Gulf hypoxia.

The programs mentioned above can be categorized into one of two groups: land retirement programs and “working” land programs. Both the CRP and WRP are examples of land retirement programs, since landowners receive payments in exchange for taking land out of active agricultural production and putting the land into perennial grasses, trees, or wetlands restoration. In contrast, EQIP and the CSP are examples of working land programs whereby landowners or producers receive payments to cover part or all of the costs of making changes in conservation practices or management decisions on their land that remains in agricultural production. Some research has addressed the cost-effectiveness of working land programs versus land retirement programs. For example, Feng et al. (2006) found that a cost-effective allocation of resources to sequester carbon in agricultural soils favors working land (via conservation tillage subsidies) over land retirement (via payments to retire land and plant it in perennial grasses). It is important to note, however, that this study focused on stylized working land and land retirement programs rather than attempting to address the cost-effectiveness of existing conservation programs as actually implemented.

The existing working land and land retirement programs are implemented with features that likely affect the cost-effectiveness of the programs for achieving environmental gains in different ways. For example, the CRP uses an EBI that favors admitting land into the program that achieves environmental benefits at relatively low costs. All else equal, this component of the program will improve its cost-effectiveness. In contrast, the CSP provides payments for ongoing stewardship of farmers so that program expenditures are used to reward past behavior rather than to change existing behavior. This, all else equal, will reduce the program cost-effectiveness for achieving environmental gains. The lack of competitive bidding and clear targeting also reduces the cost-effectiveness of this program. Finally, it is worth noting that targeting and competitive bidding were explicitly disallowed in the EQIP program during its most recent reauthorization. Again, this will reduce its cost-effectiveness.

4.4.3 Emissions and Water Quality Trading Programs

Emission trading is a regulatory approach that sets a maximum allowable level of overall emissions and then allows sources to exchange pollution allowances. A properly structured trading program can reduce the costs of achieving emission standards by allowing the flexibility necessary to focus pollution reductions on sources that are less expensive to control. In theory, a broad-based emissions trading program could help to reduce the air and water contributions of nutrients to the NGOM. Water quality trading is simply the name given to the extension of emissions trading to achieving water quality objectives.

In a recent survey of the programs to support water quality trading in the United States, Breetz et al. (2004) identified 40 water trading initiatives and an additional six state policies with specific programs related to water quality trading. USEPA has supported these programs (USEPA, 2004a) and has produced explicit policies related to their implementation. Many states and regions also have explicit policy guidance. However, the effectiveness of these programs appears to have been quite limited as very few trades are actually occurring. Further, little evidence of environmental improvement associated with these programs exists (Breetz et al., 2004).

A key problem with these programs is the lack of a required water quality improvement necessary to generate adequate demand for credits (King, 2005). To achieve "cap and trade," an effective cap is necessary. A cap could come from a tight enough cap on point sources such that they would find it cost-effective to purchase credits from agricultural nonpoint sources. Alternatively, the cap could be extended to agricultural sources. While some have conjectured that the Total Maximum Daily Load (TMDL) program may eventually play this role, there is no current mandate for agricultural sources to restrict nutrient runoff. Also problematic are a range of restrictions on allowable trading, such as requirements that a particular baseline set of conservation practices be in place with credits accruing only for additional conservation activity.

While trading could be a significant contributor to cost-effective nutrient control, the necessary institutions for water and/or air emissions trading to be an effective policy instrument are not broadly in place. In addition to clear and enforceable limits on emissions or water quality contributions (from point and/or nonpoint sources), enforceable rules concerning trading ratios, liability when standards are not met, monitoring, etc. must be established before these markets can flourish. Ideally, a trading program to address NGOM hypoxia would be broad based and include highly diverse sources (such as air deposition and many agricultural nonpoint sources) to maximize the potential for cost savings.

4.4.4 Agricultural Subsidies and Conservation Compliance Provisions

US farmers have been the recipients of farm payments for decades. These payments support prices and/or income, especially of farmers growing bulk commodities such as corn and soybeans. Economic theory suggests that, all else equal, such payments will increase the intensity and acreage of farming, possibly resulting in increased water quality problems. Research by Reichelderfer (1985) provided empirical evidence that these payments encourage crop production on highly erosive land. Likewise, a recent study from USDA's Economic Research Service (Lubowski et al., 2006) quantified the effect of one major program, subsidized crop insurance, on the location and acreage of cropland and its environmental effects. Lubowski et al. (2006) estimated that about a million hectares (2.5 million acres) were brought into production as a result of the program and that these lands are more vulnerable to erosion, are more likely to include wetlands, and have higher levels of nutrient losses than average.

To some extent, USDA's conservation programs (see Section 4.4.2) exist to counteract the "perverse effects" or unintended consequences of its crop subsidies inasmuch as government financial support has encouraged farmers to choose commodity crops that require more fertilizer, maximize yield without regard to soil and water quality consequences, and cultivate marginal land. Restructuring or eliminating existing subsidies could serve to mitigate some of these perverse effects (e.g., by shifting subsidies to reward less fertilizer-intensive crops as well as by requiring, as a condition of receiving subsidies, certain conservation practices).

Taheripour et al. (2007) provided additional evidence on this point. First, their model suggests that removal of all crop subsidies would reduce nitrogen pollution by 8.5% and that the reduced need for distortionary income taxes to support these subsidies could increase social welfare by $1.2 billion. Further, they found that tax-neutral policies to achieve nitrogen reduction can generate significant double dividends (a double dividend refers to a situation where a policy not only internalizes an externality but also reduces the deadweight losses associated with distortionary taxation, such as an income tax). They provide an estimate of the magnitude of the double dividend for a range of nitrogen-reduction goals and policy approaches including a nitrogen tax, a nitrogen reduction subsidy, a tax on output, and a combined output tax and nitrogen reduction subsidy and find that a double dividend from these instruments can be significant.

While environmental improvements associated with agriculture have largely been pursued via cost-share or subsidy programs, one significant regulatory approach has been the implementation of environmental compliance provisions that require farmers who receive farm program payments (including price support and income support) to undertake some environmental performance practices. Specifically, in the 1985 Food Security Act, conservation compliance provisions required owners of highly erodible land (a categorization of land based on its slope and soil type) to implement soil conservation plans, and a "swampbuster" provision disallowed payments to go to farmers who converted wetlands to crop land. Claassen et al. (2004) estimated that up to 25% of the reduction in soil erosion that occurred between 1982 and 1997 was attributable to conservation compliance. Many believe these gains could have been higher if there had been stronger enforcement of the mechanism. While no direct estimates are available of the increased benefits that could come from more enforcement, there is evidence of very limited reporting and penalizing of violations (Claassen, 2000).

Claassen et al. (2004) assessed the prospect for reducing nutrient losses from the Mississippi River basin by extending compliance requirements to nutrient management. They used "nutrient management" to refer to the range of activities related to the timing and level of fertilization decisions that best minimizes soil nutrients in excess of crop needs at any point in time. They noted that the ideal set of nutrient-management practices will vary considerably across farms and regions and that the costs of these activities will also vary notably across this space. Using data from the EQIP program, they summarized the distribution of incentive payments needed to induce willing adoption of nutrient management practices as defined under EQIP. For the Heartland region (ERS Farm Resource Region), the average annual incentive payment is about $7/ac, and 95% of the payments are $12/ac or less.

While these data provide an excellent starting point for assessing the cost-effectiveness of nutrient management methods addressing local water quality and NGOM hypoxia, several additional pieces of information would be needed for a full assessment. First, these costs represent the compensation needed for those farmers who have already adopted practices under the EQIP program; those who have not adopted are likely to have at least as high costs, possibly substantially higher. In this regard, these costs could be viewed as a lower bound. Second, these costs are specific to the EQIP requirements for nutrient management. Whether these requirements are effective enough to yield substantial off-site benefits is not addressed. Nonetheless, based on this cost assessment and a comparison with the annual commodity program payments farmers typically receive, Claassen et al. (2004) concluded that substantial nutrient management could occur with extension of conservation compliance provisions to nutrients.

Claassen et al. (2004) also considered whether buffer practices could be induced under conservation compliance provisions. They included riparian buffers, filter strips, grassed waterways, and contour grass strips in their discussion of buffer practices. To assess the costs of these practices and how they vary across locations, they looked at information on producers' willingness to accept compensation for adoption of the practices observed for continuous CRP priority areas. Owners of these lands received an average payment of about $90/year in addition to 50% cost share for installation of the buffer practice. Based on this analysis, as an example, Claassen et al. (2004) computed the annual costs per area for a filter strip and concluded that, in many cases, this payment would be below the average subsidy received by producers, thereby suggesting that buffer practices might also be successfully adopted under nutrient compliance provisions.

Finally, Claassen et al. (2004) noted that conservation compliance provisions are likely to have few transaction costs relative to other policies (although enforcement costs would need to be considered) and require very low budgetary outlays beyond the payments that are already provided for commodity or insurance programs. Claassen et al. (2004) also argued that conservation compliance requirements have been relatively cost-effective due to the flexibility with which they can be implemented. Producers in different regions of the country, with differing soil and weather conditions, can meet their compliance obligations with different practices. This flexibility means that the most appropriate technologies can be used for the location of the practice.

4.4.5 Taxes

The use of a per unit tax to internalize the costs of externalities of production is well known to be highly cost-effective when the tax is placed directly on the externality generating activity; these "Pigouvian" taxes are the equivalent of placing the appropriate price on the pollutant (Baumol and Oates, 1988). Taxes can be a powerful market signal, communicating the need to change behavior, Baumol and Oates (1988) demonstrated that subsidies (essentially just negative taxes) can also be designed that provide the equivalent market signals for changes in behavior. This argument is often used to support the design of environmental programs that pay participants for the provision of environmentally friendly practices rather than using taxes to change behavior. A potentially important exception to this equivalence can occur when the provision of a positive payment induces entry into the farming sector generating production on otherwise unprofitable lands. This possibility was addressed in Section 4.4.4 in the context of general agricultural subsidies and conservation compliance.

A tax directly on an input into production that is highly correlated with the pollutant can be an efficient second-best policy. The possible use of a nitrogen fertilizer tax was considered in Doering et al. (1999) and found to be as cost-effective as any of the policies they considered (they note that the initial incidence falls on farmers). Fertilizer taxes already exist in some states but are set at much smaller levels than those studied by Doering et al. (1999). The inelastic demand for fertilizer (Denbaly and Vrooman, 1993) means that the magnitude of taxes needed to induce behavioral change would likely be large.

The incidence of a tax (and thus determination of who pays the costs) is likely to fall on farmers and consumers of food products made from crops that use fertilizer. In contrast, the incidence of conservation program payments is largely on taxpayers. Finally, it is important to note that tax instruments will be more efficient the more broadly they are applied to the various nutrient sources identified as pollutant contributors; so ideally a tax would be applied to all nutrient sources rather than singly to fertilizer.

4.4.6 Eco-labeling and Consumer Driven Demand

The idea that environmentally friendly producer behavior can be induced by consumer demand is one basis for eco-labeling and certification programs. Dolphin safe tuna (Teisl et al., 2002) and organic fruits and vegetables (Loureiro et al., 2001) are two successful examples. Research analyzing the effectiveness of eco-labeling suggests some promise.

Thogersen (2002) summarized three schemes, all implemented in Europe, that have been credited with significant reductions in emissions from heating appliances and paint solvents (the German "Blue Angel" brand) and reductions in pollutants from paper production and household chemical and laundry emissions (the Swedish "Good Environmental Choice" label and the Nordic "Swan" label). Although not specific to a particular product, Clark and Russell (2005) noted that several studies of the Toxic Release Inventory have shown that information can affect firms' choices.

Could consumer-driven demand affect the changes in land-use and agricultural management necessary to contribute notably to nutrient flows into the Gulf? This approach would require the labeling of food and fiber products made from agricultural outputs in the MARB to indicate that they were produced in such a way as to reduce or eliminate nutrient contributions to hypoxia. Consumers would then need to respond to this labeling by purchasing products, presumably at a higher cost, in adequate quantity to change the market behavior. Given that much of the grain produced in the Corn Belt is used for livestock feed and not directly traceable to its field of origin, it will be difficult to distinguish products that were produced with "hypoxia -friendly" production practices from those that were not. It is not clear that labeling can credibly be produced without significant government involvement and expense (Crespi and Marette, 2005). Nor is it clear that consumer response would be adequate to drive changes in production practices, even if the labeling challenges could be overcome. One area in which labeling may prove effective is in animal agriculture, where the tracking of an individual unit from producer to final consumer is more straightforward.

Key Findings and Recommendations

Voluntary agreements with no accompanying economic incentives are not likely to be adequate to obtain significant reductions in N and P. While there may still be some low-cost conservation practices that can be implemented in some locations (better "crediting" for manure spreading for example), nutrient reductions that face agricultural producers with costly trade-offs cannot be expected without strong economic signals. These economic incentives can take many forms: conservation payments such as those in many current agricultural conservation programs, taxes, restructuring or removal of subsidies (such as conservation compliance provisions).

Water quality trading programs have not yet demonstrated the ability to improve environmental performance and/or reduce costs of meeting environmental targets primarily due to an absence of effective emissions restrictions. However, with clearer water quality improvement mandates and more flexible rules for trading, these programs could develop into cost-effective instruments.

Numerous studies have demonstrated that existing incentive-based conservation programs, specifically the CRP, WRP and EQIP, have provided significant environmental benefits. However, these programs can be much more cost-effective with additional targeting and competitive bidding mechanisms. Given the menu of existing programs, it is possible to reduce hypoxia and protect water quality in the MARB without significant new government funding, although the distributional consequences of the various approaches will differ. Based on these findings, the Study Group offers the following recommendations.

- To achieve N and P reductions from agricultural sources of the magnitude needed to affect hypoxia, economic incentives are needed to induce adequate adoption of conservation practices. These incentives can take many forms: conservation payments, taxes, and/or restructuring of existing farm subsidy and compliance requirements.

- To maximize the N and P reductions achieved with federal and state conservation dollars (e.g., CRP, WRP and EQIP), targeting and competitive bidding mechanisms are needed so that lands enrolled in these programs achieve maximum environmental benefits at lowest cost. Strategically placed wetlands in the upper Mississippi River basin could serve as effective nutrient sinks. Research has demonstrated that the local co-benefits are large enough, in and of themselves, to justify restoring these wetlands. The additional benefits associated with reduction in Gulf hypoxia reinforce the conclusion of the desirability of wetlands restoration.
- Water quality trading programs hold promise, but, without enforceable caps (water quality standards), these programs cannot be expected to achieve much nutrient reduction.
- To minimize the adverse effects of existing agricultural subsidy programs, conservation compliance requirements that target reductions in nutrients could be very cost-effective, but only with adequate enforcement.
- To select policies and programs with maximum economic efficiency, all co-benefits should be considered regardless of which policy tools are used. For example, since wetlands provide valuable habitat and flood control in addition to water quality benefits, there may be instances in which it is desirable to control nutrients by restoring wetlands, even if it is less costly to reduce nutrients by managing croplands.

4.5 Options for Managing Nutrients, Co-benefits, and Consequences

4.5.1 Agricultural Drainage

The *Integrated Assessment* reports identified several research needs related to agricultural drainage. Brezonik et al. (1999) emphasized the importance of agricultural drainage in nutrient transport from cropland and identified increased spacing of subsurface drainage tile and controlling water table levels (controlled drainage) among those practices that could potentially reduce nitrate losses from cropland. Mitsch et al. (1999) noted that controlled drainage was not widely practiced in US Corn Belt and that most of the research on controlled drainage had been conducted in more southern climates.

4.5.1.1 Alternative Drainage System Design and Management

Relatively few field studies have addressed the effects of subsurface drain depth and spacing on N losses from cropland. Overall, results suggest a trend of decreased subsurface flow and decreased N loss at wider tile spacing or decreased tile depth. Reported reductions in nitrate export are primarily due to reductions in the volume of flow rather than reductions in nitrate concentration. Drain flows and N loss can be affected by both drain spacing and depth (Hoffman et al., 2004; Kladivko et al., 2004; Skaggs et al., 2003, 2005), and use of drainage intensity (Skaggs et al., 2005) normalizes some of the variability in results of drainage spacing studies. Drainage intensity increases with deeper tile depths and closer tile spacing. Research suggests that reducing drainage intensity by either shallower tile depth or wider tile spacing will reduce subsurface flow and nitrate loss. However, adjustments in tile spacing and depth are only possible when drainage systems are being installed, and the Corn Belt is already extensively drained. As these systems are replaced, repaired, and upgraded over the next few decades, there will be opportunities to consider alternative drainage designs to minimize nutrient losses. In the meantime, there may be opportunities to achieve similar benefits by retrofitting existing drainage systems with control structures that allow some management of subsurface drainage.

Drainage management (controlled drainage) is currently an area of active research and development (http://extension.osu.edu/~usdasdru/ADMS/ADMS index.htm). Research suggests that drainage management could reduce nitrate transport from drained fields by 30% for regions where appreciable drainage occurs in the fall and winter (Cooke et. al., 2008). Although water table management could potentially alter nitrification and denitrification reactions, reported reductions in nitrate export with controlled drainage are primarily due to reductions in the volume of flow rather than reductions in nitrate concentration. Some uncertainty arises from difficulties in closing water balances (and therefore N balances) in field studies, and an unknown amount of subsurface flow reduction could be due to lateral seepage and/or increased surface runoff (Cooke et al., 2008). Simulation studies predict increased surface runoff when higher water tables are maintained using controlled drainage (Singh and Helmers, 2006; Skaggs et al., 1995), suggesting a potential tradeoff between reduced subsurface drainage and increased surface runoff. Although raising the water table can decrease the volume of infiltrating water entering drainage tile, higher water tables can also increase surface runoff resulting in increased erosion and loss of particulate contaminants such as soil bound phosphorous.

Controlled drainage requires relatively flat and uniform topography, and slopes of less than 0.5 or 1% are recommended (Cooke et al., 2008; Frankenberger et al., 2006). Concerns for erosion and surface runoff increase with increasing slope, and slopes greater than 0.5–1% can require an impractical number of control structures. There has been speculation that new technologies could make the practice economically feasible at slopes of 2% or more, but this would raise even greater concerns over surface runoff. Although tile drainage is widespread throughout the Corn Belt, it is not clear what portion of this tile drainage can be retrofitted with structures for controlled drainage. A first approximation might be an estimate of the fraction of tile-drained lands with slopes less than 0.5–1%, but this approach requires higher resolution topography than is generally available in the Corn Belt. These estimates are available for a few large drainage districts in north central Iowa for which very high resolution topography were developed. Although 50–75% of the cropland in these drainage districts is tile drained, only about 10% has a slope less than 1% and only about 3% has a slope less than 0.5% (Matt Helmers, Iowa State University, Ag Drainage Website, http://www3.abe.iastate.edu/agdrainage). These results suggest that controlled drainage may be applicable to a relatively small fraction of tile-drained land in Iowa, but this may not be representative of other regions of the Corn Belt. Based on STATSGO soils data, Illinois, Indiana, and Ohio may have twice as much cropland suitable for controlled drainage as Iowa (Dan Jaynes, National Soil Tilth Lab, Ames, IA). High-resolution topography could provide a much better basis for this assessment.

4.5.1.2 Bioreactors

Denitrification bioreactors have been installed in the field as treatment systems for tile drain effluent (Van Driel et al., 2006) and as denitrification walls (a trench filled with carbonaceous material to intercept subsurface flow) (Robertson et al., 2000; Schipper et al., 2004, 2005; Schipper and Vojvodic-Vukovic, 1998, 2001). Bioreactors on tile drains are typically bypassed during high flows and "are most usefully applied in the treatment of baseflows rather than peak flows." Current knowledge indicates that denitrification walls are effective for at least 5–7 years with little or no loss of nitrate removal capacity (Robertson et al., 2000; Schipper and Vojvodic-Vukovic, 2001). A variety of materials such as corn stalks, wood chips, and sawdust are potential organic amendments to enhance denitrification in bioreactors. Continued research is needed to determine whether denitrification bioreactors could be installed around lateral tile drain lines and whether this would be technically and economically feasible. Future redesign of tile drain systems may include integrated denitrification enhancements around tile lines and at the outlets of smaller tile lines.

Key Findings and Recommendations

Alternative drainage designs with reduced drainage intensity due to shallower tile depths and/or wider tile spacing could significantly reduce nitrate losses but can be expected to increase surface runoff and losses of particulate contaminants. Controlled drainage could significantly reduce nitrate losses where appreciable drainage occurs in the fall and winter but can be expected to increase surface runoff and losses of particulate contaminants. Controlled drainage is most appropriate for areas having slopes of less than 0.5–1%, and it is not clear what fraction of tile-drained lands are suitable for application of controlled drainage. In some areas, slope could seriously constrain applicability of the practice. Bioreactors can significantly reduce nitrate concentrations but typically must bypass peak flows during which much of the nitrate load is transported. Based on these findings, the Study Group offers these recommendations.

- Additional research is needed to evaluate topographic constraints on the applicability of controlled drainage including developing high-resolution topography for the Corn Belt.
- Additional research is needed to fully characterize water and nutrient balances for alternative drainage design and management, most critically using small watershed-scale studies (less than 2,500 hectares or about 10,000 acres) to document effects when scaled up.
- A strategy for implementation of alternative drainage design or management should be developed that includes consideration of potential trade-offs between reduced nitrate loss through tile drains and increased P loss through surface runoff.

4.5.2 Freshwater Wetlands

If wetlands are to serve as long-term "sinks" for nutrients, reductions in nutrient loads must reflect net storage in the system through accumulation and burial in sediments or net loss from the system, for example through denitrification or vegetation removal. The effectiveness of wetlands in reducing N export from agricultural fields will depend on the magnitude and timing of NO_3 loads and the capacity of the wetlands to remove NO_3 by denitrification. In contrast to NO_3, gaseous losses of P are insignificant, and sediment accretion of bound inorganic P and unmineralized organic P is the primary mechanism by which wetlands serve as long-term P sinks. With the exception of P associated with suspended solids, wetlands are generally less effective at retaining P than at removing NO_3 (Reddy et al., 1999).

4.5.2.1 Nitrogen

The effectiveness of wetlands in NO_3 reduction is a function of hydraulic loading rate, hydraulic efficiency, NO_3 concentration, temperature, and wetland condition. Of these, hydraulic loading rate and NO_3 concentration are especially important for wetlands intercepting nonpoint source loads. Hydrologic and NO_3 loading patterns vary considerably for different landscape positions and different geographic regions. The combined effect of variation in land use, precipitation, and runoff means that loading rates to wetlands receiving nonpoint source loads can be expected to vary by more than an order of magnitude and will, to a large extent, determine NO_3 loss rates for individual wetlands.

Mitsch et al. (2005a) examined NO_3 retention in Mississippi River basin wetlands receiving nonpoint source NO_3 loads either directly or through diversion of river water. Their study extended the earlier analysis of Mitsch et al. (1999) to include additional wetlands and to include wetlands outside the agricultural regions of the Corn Belt. They found that 51% of the NO_3 mass reduction by the wetlands examined could be explained by a nonlinear regression based on annual mass load of NO_3 per area of wetland. However, when the analysis is restricted to Corn Belt

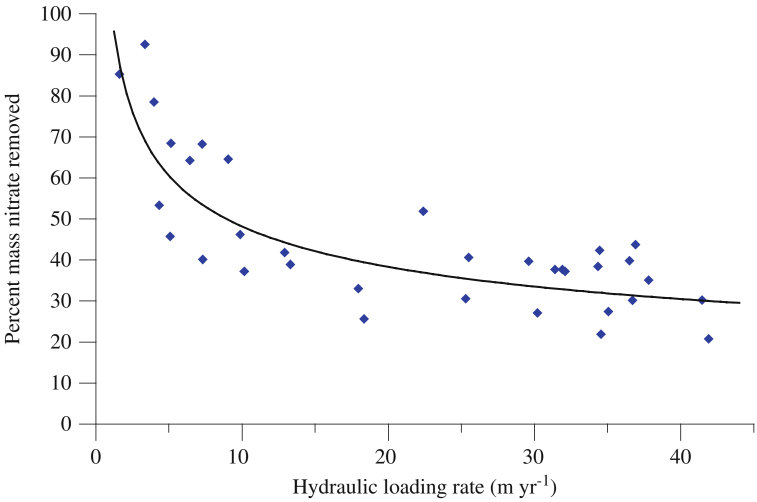

Fig. 4.2 Percent mass nitrate removal in wetlands as a function of hydraulic loading rate. Best fit for percent mass loss = 103 * (hydraulic loading rate)$^{-0.33}$ ($R^2 = 0.69$). Adapted from Crumpton et al. (2006, 2008)

wetlands that receive seasonally variable water and nutrient loads (i.e., subjected to nonpoint source loading regimes), the relationship is much weaker (Crumpton et al., 2006, 2008). Based on 34 "wetland years" of available data (12 wetlands with 1–9 years of data each) for sites in Ohio (Mitsch et al., 2005a; Zhang and Mitsch, 2000, 2001, 2002, 2004), Illinois (Hey et al., 1994; Kovacic et al., 2000; Phipps, 1997; Phipps and Crumpton, 1994), and Iowa (Crumpton et al., 2006; Davis et al., 1981), percent mass NO_3 removal is much more closely related to hydraulic loading rate (HLR) (Fig. 4.2, $R^2 = 0.69$) than to mass loading rate ($R^2 = 0.22$).

Hydraulic loading rate explains relatively little of the variability in NO_3 mass removal, which can vary considerably more than percent NO_3 removal among wetlands receiving similar hydraulic loading rates. However, much of the variability in mass NO_3 removal can be accounted for by explicitly considering the effect of HLR and flow-weighted average (FWA) NO_3 concentration (Crumpton et al., 2006, 2008). For the wetlands in Fig. 4.2, mass NO_3 removal rate can be predicted as the product of percent removal (estimated as 103 * HLR$^{-0.33}$) and mass load (estimated as HLR * FWA). This simplifies to the function [mass removal in kg N/ha-year = 10.3 * (HLR in m/year)$^{0.67}$ * FWA NO_3 concentration in g N/m^3] and explains 94% of the variability in mass NO_3 removal for the wetlands considered here (Fig. 4.3). The isopleths on the function surface in Fig. 4.3 represent the combinations of HLR and FWA that can be expected to achieve a particular mass loss rate and illustrate the benefit of targeting wetland restorations in areas with higher NO_3 concentrations. The wetlands examined by Mitsch et al. (2005a) had a median loading rate of 600 kg NO_3-N/ha-year, at which they predicted losses of 290 kg NO_3-N/ha-year. This mass loss rate is near the lower mass loss isopleth of Fig. 4.3 as would be expected for

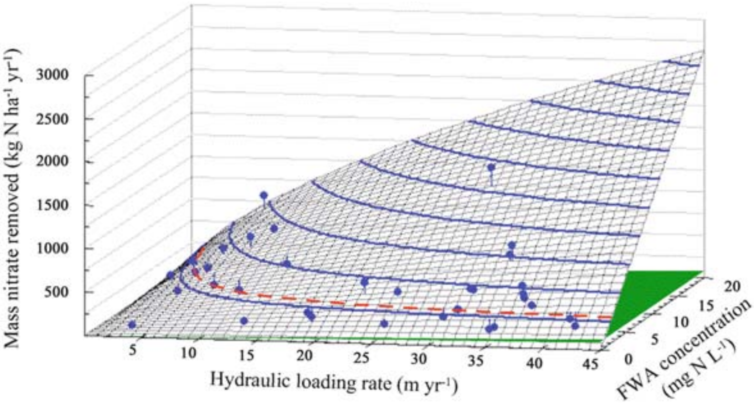

Fig. 4.3 Observed NO_3 mass removal (*blue points*) versus predicted NO_3 mass removal (*blue surface*) based on the function [mass NO_3 removed $= 10.3^*(\mathrm{HLR})^{0.67} *$ FWA] for which $R^2 = 0.94$. *Blue lines* are isopleths of predicted mass removal at intervals of 250 kg/ha-year. The *dashed, red line* represents the isopleth for mass removal rate of 290 kg/ha-year suggested by Mitsch et al. (2005a). The *green plane* intersecting function surface represents organic N export. Adapted from Crumpton et al. (2006, 2008)

either low FWA concentrations at moderate to high HLRs or higher FWA concentrations at lower HLRs. Half of the wetlands considered by Mitsch et al. (2005a) had NO_3 concentrations below 3 mg N/l. NO_3 concentrations in tile drainage water commonly exceed 10–20 mg N/l (Baker et al., 1997, 2004, 2008; David et al., 1997; Sawyer and Randall, 2008). The greatest benefit of wetlands for mass NO_3 reduction will be found in those extensively row-cropped and tile-drained areas of the Corn Belt where NO_3 concentrations and loading rates are highest. For these areas, NO_3 mass removal rates could be several times higher than predicted by Mitsch et al. (1999, 2005a).

Total and organic N data were available for about half of the wetlands represented in Fig. 4.3. All of these wetlands were sinks for total N, but most were net producers of organic N, although in comparatively small amounts (and none were net producers of NH_4). On an average, FWA organic N discharged from the wetlands increased by approximately 0.2 g N/m^3 (range from <0 to 0.3 g N/m^3) relative to incoming concentrations, with no relation to HLR or NO_3 concentrations. The mass export of organic N was small compared to NO_3 removal and had relatively little impact on reductions in total N, especially at higher NO_3 concentrations. For comparison to mass NO_3 loss, mass organic N export can be estimated as the product of HLR and the increase in FWA organic N and is represented by the green plane intersecting the function surface in Fig. 4.3. At elevated NO_3 concentrations, wetlands are nearly as effective in reduction of total N as in reduction of NO_3. At very low NO_3 concentrations, organic N production could equal NO_3 removal, in which case wetlands would not function as total N sinks.

There is some concern over increased N_2O emissions in wetlands exposed to high nitrate loads, and N_2O emissions do increase in wetlands at elevated nitrate levels. However, N_2O accounts for a very small fraction of N removal in wetlands receiving non point source nitrate loads, and N_2O emission rates from these systems are very low (Hernandez and Mitsch 2006; Paludan and Blicher-Mathiesen 1996; Stadmark and Leonardson, 2005). N_2O emission accounted for only 0.3% of total N loss in wetlands receiving river flows with elevated nitrate levels (Hernandez and Mitsch, 2006), and less than 0.13% of total nitrate loss in a wetland recharged by groundwater with elevated nitrate levels (based on maximum flux rates reported by Paludan and Blicher-Mathiesen, 1996). N_2O emission rates in wetlands receiving nonpoint source nitrate loads average around 1 μmol N_2O m^{-2} h^{-1} (Hernandez and Mitsch, 2006; Paludan and Blicher-Mathiesen, 1996), which is very similar to rates reported for cultivated crops in the Midwest (1–2 μmol N_2O m^{-2} h^{-1}) (Grandy et al., 2006; Parkin and Kaspar, 2006). The available research suggests that wetlands restored on formerly cultivated cropland for the purpose of nitrate removal would have little or no net effect on N_2O emissions.

4.5.2.2 Phosphorus

P removal in wetlands is controlled by three sets of processes: (1) sorption or release of P by existing sediments, (2) accumulation of P in new biomass, and (3) accumulation of P associated with the formation and accretion of new sediments/soils (Reddy et al., 2005). Existing sediments will have a finite capacity for sorption of P, determined in part by Al and Fe content in acid soils and by Ca and Mg content in alkaline soils. There will also be a finite capacity for the accumulation of P in new biomass. Of the three sets of processes, only the last contributes to long-term, sustainable P retention by wetlands: the accumulation of bound inorganic P and unmineralized organic P associated with the formation and accretion of new sediments and soil.

P sorption on both antecedent and newly accreting wetland soil is largely controlled by Fe, Al, and Ca. Reducing conditions found in wetlands may decrease sorption of P as insoluble complexes formed with Fe^{+3} are released upon reduction to Fe^{+2}, solubilizing the P (Patrick et al., 1973). High S levels may enhance P flux from soils due to the binding of iron by sulfides (Bridgham et al., 2001; Caraco et al., 1989). Alkaline wetland soils are more conducive to P sorption than acidic wetland soils due to the presence of Ca in the alkaline wetland soils and the formation of insoluble Ca-bound P (Bruland and Richardson, 2006; Richardson, 1999). These two studies indicate that wetlands developed on soils rich in calcite and exchangeable Ca are likely to be more effective sinks for P under the reducing conditions necessary for denitrification. More research is needed to understand (1) the effects of wetland creation such as is being done in the upper Mississippi River basin and (2) whether wetlands created/restored on Mollisols will be effective P sinks due to formation on Ca–P complexes in addition to sedimentation and SOM formation. Bruland and Richardson (2006) determined that marshes with a higher soil P sorption index (amount of P sorbed by soil from a phosphate solution in 24 h incubation) would be the best P sinks and that specific marshes could be targeted based on this index. It is important to remember, however, that antecedent soils of restored wetlands have a finite P retention capacity. The long-term sustainable capacity of these systems to retain P is determined primarily by the accumulation of

P associated with the formation and accretion of new sediments and soils. Studies of wetlands constructed to intercept nonpoint source nutrient loads in the MARB confirm the importance of sediment accretion for P retention (Anderson et al., 2005; Mitsch et al. 2005b) but also demonstrate that wetlands can become a P source if sediments are remobilized (Mitsch et al., 2005b). Most of the MARB studies represent recently constructed wetlands, and the long-term sustainable capacity of these systems to reduce P loadings is unclear.

Wetlands created, enhanced, and restored for N removal could also function for P removal, but limits to sustainable P removal must be recognized. Both NO_3 and P removal in wetlands will be enhanced by longer retention times and accretion of organic rich sediments. Long-term solutions for P load reduction in the MARB will likely depend more on reduction in sources than will long-term N load reduction. It will be important to manage restored wetlands so they do not become long-term sources of P after nonpoint sources of P have been reduced.

Key Findings and Recommendations

As concluded in the *Integrated Assessment*, wetlands can be very effective in NO_3 removal. Recent data, though limited, support the *Integrated Assessment*'s conclusion that N_2O evolution from wetlands restored as NO_3 sinks would be a low percentage of total denitrification. Wetlands receiving significant nonpoint source NO_3 loads at moderate-to-high NO_3 concentrations export comparatively small amounts of organic N and are nearly as effective in reduction of total N as in reduction of NO_3. This situation is less true for wetlands receiving loads at low NO_3 concentrations. Hydraulic loading rate and NO_3 concentration are especially important determinants of NO_3 removal rates in Corn Belt wetlands. Additional information is needed on created, restored, and enhanced wetlands including long-term monitoring for total N and P retention. Based on these findings, the Study Group offers the following recommendations.

- Wetland restoration should be evaluated for its full range of benefits.
- For greatest basin-wide reduction in nitrate load, wetland restorations should be targeted in those extensively row-cropped and tile-drained areas of the Corn Belt where nitrate concentrations and loading rates are highest and sized based on expected hydraulic loading rates and load reduction goals. For these areas, nitrate mass removal rates could be several times higher than previously predicted.
- Although limits to sustainable P removal by wetlands must be recognized, wetlands restored for N removal should be managed for P retention as well.

4.5.3 Conservation Buffers

Conservation buffer practices include riparian buffers (forests and herbaceous cover), field borders, filter strips, contour buffer strips, grass waterways, windbreaks, hedgerows, and other practices. They are part of the suite of conservation practices that are applied by farmers to achieve productivity, stewardship, and environmental quality goals. Conservation buffers differ from other conservation practices in that they will require long-term set aside of critical lands from continued agricultural production. Although often installed under the Conservation Reserve Program (CRP), conservation buffers differ from other uses of CRP because conservation buffers allow most land to remain in production while using critical areas as buffers for the agricultural land.

Prior analysis of nutrient control in the MARB focused on riparian forest buffers, one prominent type of conservation buffer (Mitsch et al., 1999). Studies conducted over the past decade in the Corn Belt have shown conservation buffers, especially riparian forest buffers and riparian herbaceous buffers, to be effective sinks for nutrients and sediment in landscapes with a significant portion of water moving as either surface runoff or shallow subsurface flow. If nitrate is transported from cropland primarily in tile drain flow as in much of the Corn Belt, riparian buffers and vegetated filter strips will have little opportunity to intercept nitrate loads. It is likely that if drainage management is changed to limit subsurface discharge through tile drains with concomitant increases in surface runoff and shallow water table flow, riparian buffers will be critical to achieve water quality goals.

Reduction of nitrogen by riparian buffers is generally determined by soil type, watershed hydrology (artificial drainage, groundwater flow paths, saturation), and subsurface biogeochemistry (organic matter supply, redox conditions) (Mayer et al., 2006). Control of P depends more on infiltration, surface roughness, and runoff retention. Many riparian buffers have been restored or established, but few have been studied to quantify water quality benefits. Richard Schultz, Tom Isenhart, and others developed the Riparian Management System for application in areas of the Corn Belt dominated by tile-drained systems. Modifications to the original USDA Riparian Buffer specification included integration of wetlands to intercept and remove tile drainage nitrate. Lee et al. (2000; 2003) reported rates of nutrient and sediment removal in multi-species buffer strips intercepting surface runoff in these systems. They found that switchgrass and switchgrass/woody buffers retained 50–80% of total N, 41–92% of NO_3-N, 46–93% of total P, and 28–85% of dissolved reactive P from surface runoff produced in simulated rainfall events.

Riparian herbaceous cover helps reduce sediment and other pollutants in surface runoff through the combined processes of deposition, infiltration, and dilution. Those functions are due to the cascading influence of perennial vegetation on soil quality when compared to soils under annual row crops. A series of studies on Bear Creek compared soil quality and related processes within riparian soils in a corn–soybean rotation with those soils in which perennial herbaceous vegetation had been reestablished (Schultz et al., 2004). Six years after establishment of riparian switch grass, those soils contained more than eight times the belowground biomass as adjacent crop fields (Tufekcioglu et al., 2003). As a result, soils in riparian herbaceous cover amassed up to 66% more total organic carbon in the top 50 cm (20 in) than crop-field soils (Marquez et al., 1999). This resulted in a two-and-a-half-fold increase in microbial biomass and a four-fold increase in denitrification in the surface 50 cm (20 in) of soil when compared to crop-field soils of the same mapping unit. As a result of increased soil quality, infiltration was nearly five times faster in soils under perennial vegetation than in row-cropped fields (Bharati et al., 2002). Riparian Management Systems such as those on Bear Creek are well suited to intercept increased overland flow that might be associated with changes in drainage management.

Several researchers have investigated the combined effects of these processes within riparian herbaceous vegetation and reported that sediment and nutrients in surface runoff can be reduced in the range of 12–90% compared to unbuffered crop fields (Dosskey, 2001; Lee et al., 2003). Major differences in impacts on the soil ecosystem depend upon the photosynthetic pathway of the dominant vegetation (e.g., C3 [cool-season grasses] or C4 [warm-season grasses]) in a buffer. Riparian herbaceous cover can help improve the quality of shallow groundwater, much like filter strips or riparian forest buffers. Hydrogeologic setting, specifically the direction of groundwater flow and the position of the water table in thin sand aquifers underlying the buffers, generally, is the most important factor determining buffer efficiency (Dosskey, 2001).

When applied as part of a conservation management system, the effectiveness of conservation buffers can be enhanced. There are few data on the field or landscape level effectiveness of conservation buffers applied with or without other conservation measures. Most data are from plot studies. Plot studies are inadequate, especially for studies of grass waterways (GWW), which are designed to convey overland flow from fields and stream bank restoration designed to reduce loss of sediment and sediment bound chemical from unstable banks. Because GWW are installed in areas of known water flow, they avoid problems of runoff bypassing filter strips and field borders. The few studies of GWW conducted at the field scale show that they are very effective at both runoff reduction and sediment trapping. In Germany, unmanaged grass waterways reduced runoff and sediment delivery by 90 and 97%, respectively, compared to adjacent fields with no GWW (Fiener and Auerswald, 2003). A GWW that was mowed closely was less effective, with reductions of 10 and 27% for runoff and sediment delivery, respectively. In New Brunswick, Canada, Chow et al. (1999) compared up- and downslope cultivation of potatoes and grain to the same crops with a terrace and grass waterway system. The conservation system reduced runoff by 31% and sediment delivery by 78%. On three small watersheds in the claypan soils region of Missouri, sediment and TP loss increased as the extent of GWW decreased (Udawatta et al., 2004).

There are ongoing efforts by USDA to estimate the impacts of conservation buffers on water quality in all watersheds with significant amounts of agriculture. The Conservation Effects Assessment Project (CEAP) will eventually provide model-based estimates of the water quality impacts of conservation practices in the MARB (Kellogg and Bridgham, 2003). Conservation buffers are an important component of USDA conservation programs. Table 4.4 summarizes the extent of seven major conservation buffer practices installed in the six subbasins of the MARB in federal fiscal years 2000 through 2006 (October 1999–October 2006) (M. Sullivan, personal communication, based on USDA-NRCS-Performance Results System, http://ias.sc.egov.usda.gov/prshome). An estimated 0.94 million hectares (2.31 million acres) of conservation buffers were installed in the MARB in 1999–2006. As shown, each hectare of conservation buffer treats 1 or 3 hectares of adjacent agricultural land, giving an estimated 3.46 million hectares (8.55 million acres) of agricultural land that has been treated by these six conservation buffer practices (Table 4.4).

Information on the extent of other conservation practices established from FY 2000 through FY 2006 is also available from the NRCS Performance Results System. Practices that are applied each year such as conservation tillage, residue management, and nutrient management may be reported more than once during the record period if there is a change in owner/operator, a new conservation plan is developed and associated practices are reported. There may have also been some systematic annual reporting in the early years of the record period (2000–2003) (Personal communication, Mike Sullivan, USDA-NRCS). All conservation tillage and residue management practices combined were applied on as much as 8.42 million hectares (20.8 million acres), and nutrient management was applied on as much as 7.4 million hectares (18.3 million acres) in the MARB in FY 2000–FY 2006 (Mike Sullivan, USDA-NRCS, Personal Communication, based on NRCS-PRS). Wetland creation, enhancement, and restoration were applied on 0.57 million hectares (1.42 million acres), drainage water management was applied on 756 hectares (1,867 acres), and stream bank restoration was installed on 3,155 km (1,972 mi). The values for 2002–2005 were reported in the USEPA Management Action Review Team report (MART, 2006a) and are similar to the above numbers when put on the same year basis.

Currently, no national databases allow a more detailed estimation of the environmental benefits of these conservation practices, including conservation buffers. This is the goal of the CEAP project. Estimates can be made based on acreage values, but these cannot take into account either placement or efficacy of practices. Cumulatively, conservation buffers, residue management, nutrient management, and wetlands have impacted up to 21 million hectares (51.9 million acres) of agricultural land in the MARB based on the FY 2000–FY 2006 areas of conservation practices. This area is the sum of residue management, nutrient management, conservation buffer acreage, wetland acreage and the potential land treated by conservation buffers (Table 4.4) and wetlands (assuming 3 hectares treated for 1 hectare of wetlands). In reality, conservation practices are applied as a system of practices, and it is likely that the total area treated through these practices is less than 21 million hectares (51.9 million acres). Additionally, the databases used are likely to include some duplicate reporting for the annual practices. The nutrient load reductions for these practices could be estimated based on amounts of N and P load retained. Although these would be crude estimates, they would provide numbers for comparison to the nutrient load reduction goals and provide a rough idea of where conservation programs stand relative to those goals.

Table 4.4 Areas (ha) of conservation buffers installed in the six subbasins of the MARB for FY 2000–FY2006

Subbasin	Contour buffer strips (ha)	Field border (ha)	Filter strip (ha)	Grassed waterway (ha)	Riparian forest buffer (ha)	Stream bank protection (km)	Windbreaks and shelterbelts (ha)	Conservation buffers applied (ha)
Ohio	3,362	5,441	50,617	21,346	32,497	755	794	114,832
Tennessee	196	1,914	10,724	817	10,752	418	2	26,025
Upper Mississippi	22,217	7,357	159,604	43,421	75,139	722	8,448	317,422
Lower Mississippi	165	7,541	10,274	661	56,106	503	391	75,486
Missouri	7,374	16,413	116,755	31,067	31,492	470	39,377	256,693
Arkansas White-Red	1,883	15,631	79,658	8,197	29,745	287	2,173	145,290
Sum	35,196	54,298	427,631	105,507	235,731	3,155	51,185	935,748
Area treated (ratio)	1:1	1:1	3:1	3:1	3:1	NA	3:1	
Area treated	70,393	108,595	1,710,525	422,030	942,926	NA	204,739	3,459,207

* Kilometers are shown for stream bank protection. Conservation buffers applied include areas in other practices not shown here that are cumulatively small areas compared to the practices shown. The areas treated are based on the ratios shown and assumes that each hectare of buffer treats either 1 or 3 hectares of adjacent agricultural land. Areas of practices are from Mike Sullivan, USDA-NRCS, Personal Communication, and are derived from NRCS-PRS, http://ias.sc.egov.usda.gov/prshome.

Key Findings and Recommendations

Conservation buffers and other conservation practices have affected a significant acreage of MARB cropland through existing federal, state, and private programs. The Study Group offers the following recommendations.

- Continued, new, and enhanced small watershed-based studies of suites of conservation practices as applied on farms and in agricultural watersheds are necessary. Analysis of effects of conservation buffers and other conservation practices in the MARB should be coordinated with the ongoing USDA Conservation Effects Assessment Project.
- Conservation buffers and other conservation practices in the MARB should be refocused on N and P retention with special attention given to the interactions of buffers with other practices. Environmental benefits indices should be calculated in a way as to provide extra weight for N and P retention.

4.5.4 Cropping Systems

Current cropping systems within the MARB are well established, but advances in N fertilizer production technology, innovative crop rotations, inter-seeding with cover crops, and alternative mulches or crop residues provide opportunities to improve water and nutrient use efficiency as well as to decrease leaching and runoff of nutrients and sediments. For example, inter-seeding of a leguminous cover crop within existing crop rotations could enhance N and P use efficiencies, as long as the cover crop is carefully managed. Also, greater adoption of perennial systems, which could include cellulosic production, has the potential to influence nutrient export via reduced N and P applications as well as altered water budgets. Evapotranspiration and infiltration will likely be greater with perennial than annual cropping systems, contributing to a decrease in potential runoff. Hydrologic and water quality issues related to perennials and cellulosic production are discussed in more detail in Section 4.5.9.

A continuous corn rotation typically results in annual N fertilizer applications between 150 and 250 kg N/ha (134 and 223 lb N/ac). This is a large amount of N fertilizer relative to amounts applied to other crops. Including other crops (particularly legumes) in a crop rotation usually reduces annual N fertilizer applications needed. In addition to applying less N, perennial crops, such as alfalfa or other grass mixtures, have longer effective growing seasons and are more efficient N users than annual crops, which translate to greater water use and less nitrate leaching.

Randall et al. (1997) compared tile drainage and nitrate loss for corn–soybean and corn–corn rotations to alfalfa and Conservation Reserve Program (CRP) grassland. From 770 to 905 mm (30–36 in) of tile water was recorded for the corn–corn and corn–soybean rotations from 1988 to 1993, whereas 416 to 640 mm (16–25 in) of tile water was recorded for alfalfa and CRP. Flow-weighted nitrate-N concentrations were less than 5 mg/L for alfalfa and CRP but ranged between 13 and 40 mg/L for the rotations including corn and soybean. The 4-year nitrate-N loss from continuous corn or corn–soybean rotations was 202–217 kg N/ha (180–194 lb N/ac), while for alfalfa and CRP the loss was less than 7 kg N/ha (6 lb N/ac). Similarly, Jaynes et al. (2001) showed for a corn–soybean rotation in central Iowa that even at economically optimum N fertilizer rates for corn (67–172 kg N/ha or 60–154 lb N/ac), NO_3 loss in tile drainage water increased from 29 to 43 kg N/ha (26–38 lb N/ac) with application rate. Also, a net N mass balance indicated that N was being mined from the soil at economically optimum N fertilizer rates and the system would not be sustainable (Jaynes et al., 2001).

Besides crop selection to enhance N and P removal, crop rotation also can be managed to maximize nutrient removal and minimize leaching. Together, crop selection and rotation can influence the amount of N and P in a soil profile as well as water available for nutrient leaching. As mentioned, legumes, such as alfalfa and soybean, that do not require supplemental N can effectively use or "scavenge" residual inorganic N remaining in the soil from previous crops. Some crops take up more P, and deep-rooted crops can remove N and P from subsoil horizons. For example, root development of a typical 3-year continuous corn system (maximum depths in May through September) does not always coincide with time of high NO_3 leaching potential (generally February–April). An alternative cropping system comprised of corn–winter wheat–alfalfa provided a much different root development pattern, one that should more efficiently retain N because it has deeper roots that are present most of the year (Sharpley et al., 2006b). Olson et al. (1970) found that NO_3 concentrations at a depth of 1.2–1.5 m (3.9–4.9 ft) in a silt loam soil were lower for an oat–meadow–alfalfa–corn rotation than for continuous corn when ammonium nitrate was applied to both systems. The reduction in NO_3 leaching was directly proportional to the number of years that oats, meadow, or alfalfa was grown in rotation with corn. The reduction was attributed to the combined recovery of NO_3 by shallow-rooted oats, followed by deep-rooted alfalfa (Olson et al., 1970). The potential for NO_3 leaching in such rotations is, therefore, less when compared with continuous annual monocropping systems.

Clearly, including perennial crops in a rotation, as well as conversion to perennial systems, can reduce NO_3 leaching, partly due to the fact that perennials are generally more efficient users of N than annuals. As a result, Randall and Vetsch (2005) raises a key question of whether significant reductions in nutrient (especially NO_3) loadings to surface waters are possible without changing from the predominant annual cropping system of corn–soybean rotation to a mixed system that includes perennials. While annual grain crop production is an essential component of agricultural systems in several areas of the MARB, the development of economically viable continuous cropping systems will help improve in-field nutrient use efficiency and decrease off-site loads. Additional co-benefits of perennials, such as switchgrass, are that they have the potential to accumulate large amounts of below-ground biomass and are effective in sequestering C (McLaughlin and Lszos, 2005; McLaughlin and Walsh, 1998). Costello et al., (2009) estimate that switching from corn to cellulosics for ethanol production could reduce NO_3 output from the MARB by 20%. On the other hand, corn-based ethanol production increases export of dissolved inorganic nitrogen (Donner and Kucharik, 2008).

Retirement of land through the Conservation Reserve Program has demonstrated different results for various cropping systems. For lands previously in corn, the reduction in N delivered to the Mississippi River may have been as much as 25–30 kg N/ha-year (22–27 lb N/ac-year). For soybean it would have been somewhat less, and for small grains, particularly wheat in the High Plains, smaller reductions in the range of 10 kg N/ha-year (8.9 lb N/ac-year) may have been realized (see Section 3.1.2). Where CRP has been used to establish buffers, not only are reductions from the retired lands realized, but the buffers can also be effective in reducing inputs of N and P from upslope cropland entering water courses via surface runoff and shallow subsurface flow. It should be noted, however, that most land enrolled in CRP is primarily sloping, erosive land that is not tile drained. For instance, McIsaac and Hu (2004) studying N flux in several Illinois rivers between 1977 and 1997 found that riverine N flux was about 100% of net N input for the tiled-drained region (27 kg N/ha-year or 24 lb N/ac-year). In the nontile-drained region, riverine N flux was between 25 and 37% of net N input (23 kg N/ha-year or 20 lb N/ac-year).

Key Findings and Recommendations

Cover crops and other living mulches can improve water and nutrient use efficiencies and reduce nitrate leaching. Further research and demonstration is needed in the MARB in several areas: examining the benefits of intercropping cover crops with annuals such as corn, determining if leguminous cover crops reduce fertilizer N requirements, and assessing how changes in cropping patterns can impact nutrient loss at both local and basin-wide scales. If farmers could be encouraged to switch to a rotation of perennial crops as compared to the predominant corn–soybean rotation system, significant N and P reductions would result. Based on these findings, the Study Group offers these recommendations.

- Cover, relay, and perennial crops should be considered in alternative cropping systems that will reduce nutrient loss. Cropping systems that efficiently include cover crops in grain and row cropping should also be encouraged in the Corn Belt region of MARB. This should focus on the use of fall-planted, small-grain cover crops more suited to the short growing season after harvest and cold winters of the upper Midwest.
- Where corn–soybean production systems exist and/or where it is not feasible to plant cover crops, it is even more important to encourage off-field conservation practices.

4.5.5 Animal Production Systems

4.5.5.1 System Development and Nutrient Flows

While overall production livestock numbers in the MARB have declined (see Section 3.2), there has been an intensification of operations in certain areas (see Figs. 4.4 and 4.5, and Appendix C). Farmers adopted the animal-feeding-operation (AFO) paradigm because of competitive pressures, changing marketing practices, a need to be responsive to consumer demand for quality meat products at a low cost, and declines in income from traditional grain crops in certain areas of the MARB with inherently infertile soils (Lanyon, 2005). This critical socioeconomic shift must be considered when proposing changes within the MARB that decrease the impact of AFO and manure management on nutrient export.

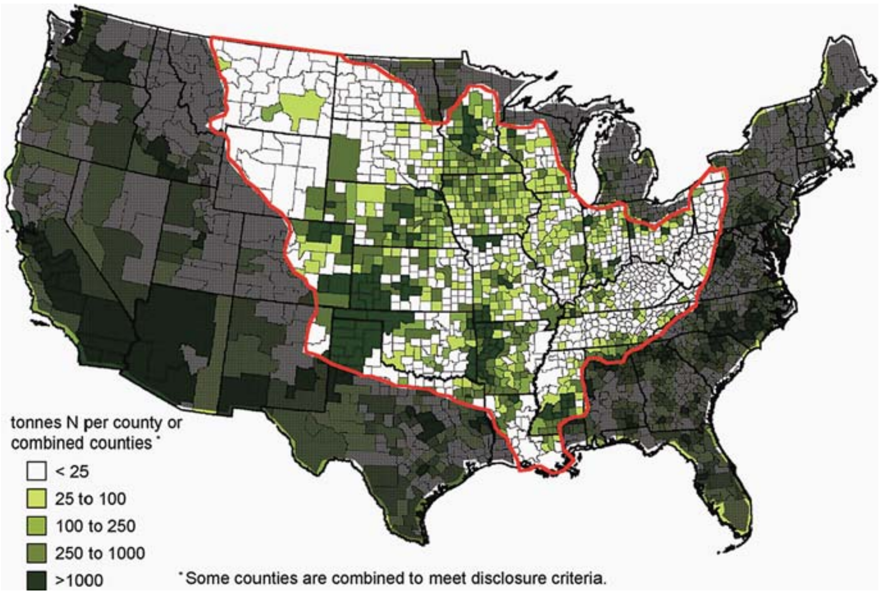

Fig. 4.4 Recoverable manure N, assuming no export of manure from the farm, using 1997 census data. Adapted from USDA (2003) with the author's permission

As a consequence of the spatial separation of crop and animal production systems, fertilizer N and P is imported to areas of grain production. The grain (harvested N and P) is then transported to areas of animal production, where inefficient animal utilization of nutrients in feed (less than 30% is utilized) are excreted as manure. This system has led to a large-scale, one-way transfer of nutrients from grain- to animal-producing areas within the MARB and dramatically broadened the emphasis of nutrient and manure management strategies from field to watershed to basin scales. For the MARB, farm-level nutrient excesses are estimated at 337 million kg N (743 million lb N) and 242 million kg P (534 million lb P) (Gollehon et al., 2001).

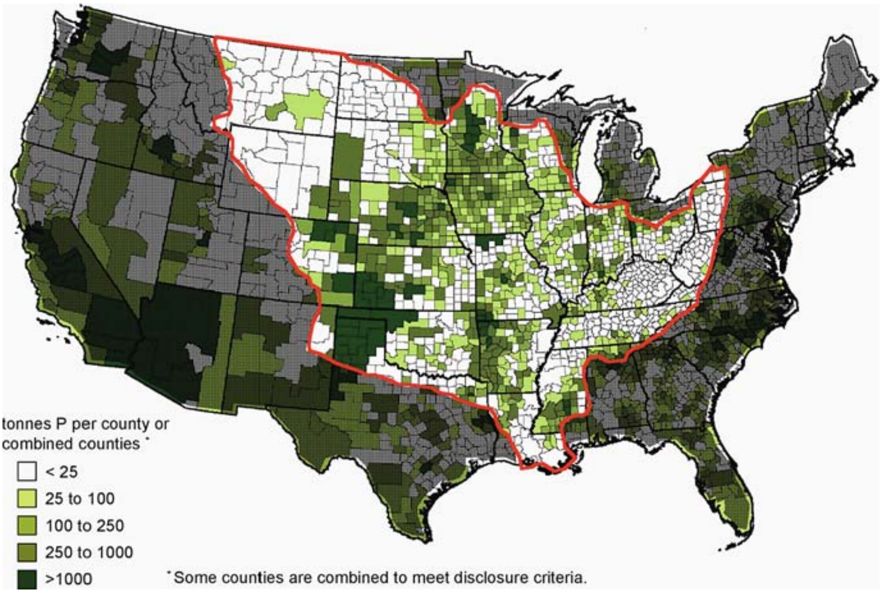

Fig. 4.5 Recoverable manure P, assuming no export of manure from the farm, using 1997 census data. Adapted from USDA (2003) with the author's permission

The land application and discharge of nutrients in manure from AFOs are regulated under the National Pollutant Discharge Elimination System (NPDES), which generally defines an AFO as an operation where livestock are confined for an extended period of time (at least 45 days in a 12-month period) and there is no grass or other vegetation in the confinement area during the normal growing season (USEPA, 2000a). This definition is intended to differentiate confinement-based operations from pasture-based operations, which are excluded from the Confined Animal Feeding Operations (CAFO) regulations. The NPDES permit is required to control pollutants at an AFO and keep them from entering surface waters. More explicitly, the USEPA (2000a) defines CAFOs as livestock operations that meet one of the following characteristics:

- Confine more than 1,000 animal units (AU), where 1,000 AUs are defined as 1,000 slaughter and feeder cattle, 700 mature dairy cows, 2,500 swine (other than feeder pigs), 30,000 laying hens or broilers if the facility uses a liquid system, and 100,000 laying hens or broilers if the facility uses continuous overflow watering.
- Confine between 300 and 1,000 AU (as defined above), and either a man-made ditch or pipe carries manure or wastewater from the operation to surface water or animals come into contact with surface water running through the area where they are confined.

These regulations are enacted at a national level, and thus, recommendations and controls on the land application or utilization of manures and their component

Table 4.5 Status of implementation of permits under the 2003 CAFO rule for states within the MARB. Data provided by USEPA Office of Wastewater Management, 2007

State	Number of CAFOs	Number of CAFOs with permits to date	Permit coverage for CAFOs under 2003 rule
Alabama	558	440	79
Arkansas	2,110	70	3
Colorado	225	33	15
Illinois	500	8	2
Indiana	584	413	71
Iowa	1,859	113	6
Kansas	476	462	97
Kentucky	150	67	45
Louisiana	150	2	1
Michigan	198	56	28
Minnesota	1,007	1,000	99
Mississippi	433	190	44
Missouri	492	492	100
Montana	TBD	75	TBD
Nebraska	1,000	303	30
New Mexico	151	47	31
North Carolina	1,222	1,200	98
North Dakota	47	0	0
Ohio	162	64	40
Oklahoma	625	163	26
Pennsylvania	462	165	36
South Dakota	369	303	82
Tennessee	129	130	101
Texas	1,204	639	53
Virginia	150	0	0
West Virginia	30	0	0
Wisconsin	161	161	100
Wyoming	51	47	92
Total	**14,505**	**6,643**	**46**

nutrients are in place at a state level in the MARB. Based on a USEPA summary of CAFO permit implementation completed in the first quarter of 2007, less than half of the CAFOs in the MARB were permitted (46%; Table 4.5). States included are in Table 4.5, if part of the state drains into the MARB. The approximate number of permitted CAFOs in the MARB is similar to the national average (44%; USEPA, 2007); but clearly, rule implementation varies among states.

4.5.5.2 Manure as a Component of N and P Mass Balances

Within the MARB, counties with the greatest excess of recoverable manure N and P (if applied on the farm where it is generated) tend to be in the western and drier areas of the basin, Arkansas, and central Minnesota (Figs. 4.4 and 4.5). Recoverable manure is defined as the portion of manure *as excreted* that could be collected from buildings and lots where livestock are held and, thus, would be available for land application. Recoverable manure nutrients are the amounts of manure N and P that would be expected to be available for land application (USDA, 2003). They are estimated by adjusting the quantity of recoverable manure for nutrient loss during collection, transfer, storage, and treatment. Recoverable manure nutrients are not adjusted for losses of nutrients at the time of land application. Where riverine N export is the greatest (upper Mississippi and Ohio River basins with tile drainage), manure N excess tends to be less. Lower Mississippi River basin states, particularly Arkansas and northern Missouri, clearly have more manure P on some farms than land area to apply it (Fig. 4.4). Although N from manure can be important in specific areas, basin-wide N loss is a result of the dominant inputs of fertilizer and N_2 fixation on tile-drained corn and soybean fields. For P, manure is a more important source, particularly on the western side of the basin (Fig. 4.5).

Large-scale consolidation has created much larger AFOs, which makes economical utilization and redistribution of manure to croplands difficult, and has profound consequences for regional nutrient transfer and management within the MARB. Furthermore, the potential for co-locating AFOs with areas of the corn production for ethanol generation may exacerbate the accumulation of manure-based nutrients in these areas. This co-location stems from the use of by-products from ethanol production (distiller's grain) as animal feed (for more information see Section 4.5.9).

4.5.5.3 Remedial Strategies

Manure is a valuable resource for improving soil structure and increasing vegetative cover, thereby improving water quality via reduced runoff and erosion potential. Manures have been historically applied at rates designed to meet crop N requirements. This has resulted in the accumulation of soil P above levels required for crop production, and a concomitant increase in the potential for N and P loss via runoff, leaching and N_2O emission within the MARB (Table 4.6; Aillery et al., 2005; Sharpley et al., 1998). In the past, separate strategies for either N or P have been developed and implemented at farm or watershed scales. The Study Group recognizes that this approach needs to change; N and P need to be managed jointly in order to improve water quality. Because of different critical sources, pathways, and sinks controlling N and P export, remedial strategies directed at only N or only P control can negatively impact the other nutrient. For example, basing manure application on crop N requirements to minimize nitrate leaching can increase soil P and enhance P losses (Sharpley et al., 1998; Sims, 1997). In contrast, reducing surface runoff losses of P via conservation tillage can enhance nitrate leaching in some cases (Sharpley and Smith, 1994).

Long-term sustainable management of nutrients in manure begins with sound feed decisions, which generally lie with the integrator in the CAFO industry rather than the individual farmer. Nutrient inputs to a farm should be matched as closely as possible with export as animal or crop products. If a farm's N and P budget is rich in imports, regardless of any other management decisions, there will be an ongoing accumulation of N and P on the farm, which in the long

Table 4.6 Estimates of manure production and N and P loss to water and air from Animal feeding operations within the Mississippi River basin. Total manure in millions of milligrams; other materials in millions of kilograms. Based on information from the 2002 US Census of Agriculture (adapted from Aillery et al., 2005)

Region of MARB	Number of operations	Total manure	N runoff	N leached	N emissions	Total N loss	P runoff
Lake States (MI, MN, WI)	52,498	62.52	32.89	0.36	164.45	198	5.58
Corn Belt (IA. IL, IN, MO, OH)	71,252	85.09	39.73	0.47	234.89	275	11.78
Northern Plains (KS, ND, NE, SD)	26,087	72.27	36.31	0.37	168.44	205	6.99
Appalachia (KY, NC, TN, VA, WV)	22,776	79.57	54.65	0.91	259.16	315	15.79
Delta States (AR, LA, MS)	12,252	19.97	8.92	0.15	62.57	72	4.47
Southern Plains (OK, TX)	10,500	49.19	21.96	0.20	119.74	142	7.72
Total	195,365	368.63	194.46	2.46	1009.26	1206	52.34

term will ultimately increase the potential for nutrient loss to water or air when manure is land applied. Nevertheless, the short-term impacts of land-applying manure or litter on nutrient loss can be reduced by the adoption of conservation practices detailed by USDA-NRCS (ftp://ftp-fc.sc.egov.usda.gov/NHQ/practice-standards/standards/590.pdf). However, conservation measures at both farm and watershed scales involves a complex suite of options, which must be customized to meet site-specific needs (for more information see Section 4.5.10 and Appendix C).

4.5.5.4 Alternative Manure Management Technologies

Reducing farm-gate inputs of N and P in animal feed presents one of the best nutrient management opportunities to effect a lasting reduction in N and P loss (Appendix C). Other measures, generally aimed at reducing the potential for N and P losses, are seen as short- rather than long-term solutions to environmental concerns. For instance, long-term monitoring of P budgets in Ohio showed that after nearly 20 years of BMP adoption and despite continually increasing soil test P levels, manure applications and timing have been managed better, resulting in more efficient use of P and reduced P loss to surface waters (Baker and Richards, 2002). Manure-related conservation practices include the following:

- manure amendments, such as alum, to reduce ammonia volatilization and sequester P in less soluble forms;
- coagulant and flocculent techniques to separate and concentrate nutrients in liquid manure systems; and
- combining manure with biosolids and woodchips to reclaim soils that have been disturbed (e.g., by mining or urban development).

As the cost of N fertilizer increases, it is clear that new markets for alternative uses or products for manure will open up. For example, on-farm and regional energy production via burning of manure is of increasing cost-effectiveness. Ash production via burning, while rich in P, will be appreciably less bulky and, thus, enable cost-effective transportation further from the source of generation. The bulky nature of manures, and resulting high cost of transportation, has always been a major limitation to more effective redistribution of N and P to nutrient deficient areas of the MARB.

Recent efforts to exclude cattle from streams as part of the Conservation Reserve Enhancement Program (CREP) were estimated to have resulted in a 32% decrease in P loadings to streams within the Cannonsville watersheds in south central New York (James et al., 2007). Thus, exclusionary programs like CREP and stream bank fencing are working to reduce nutrient loading by fencing cattle out of the stream and adjacent riparian zones. Clearly, grazing management and placement of stream bank fencing are important to minimizing watershed export of P. For instance, herd size, pasturing time, and cattle type could all be used to prioritize sites for stream bank fencing installation. In addition, field observations [such as those by James et al. (2007)] show installation of alternative watering sources do not necessarily preclude continued use of streams as a preferred water source.

The wider adoption of manure hauling that links producers with buyers will greatly enhance the sustainability of AFOs. At a state level, the Discovery Farms program is conducting research on privately owned Wisconsin farms in different geographic areas, facing different environmental challenges (see http://www.uwdiscoveryfarms.org/new/index.htm). The Discovery Farms program has been very successful at gaining farmer support in at-risk catchments in efforts to find the most economical solutions to overcoming the challenges environmental regulations place on farmers. At a watershed level, the Illinois River Watershed Partnership (see http://www.irwp.org/index.html) was established in 2005 to improve and protect water quality in the Illinois River in Arkansas and Oklahoma by working at a grassroots level with watershed citizens and other organizations.

Key Findings and Recommendations

The impacts of animal production systems are mainly expressed at a local rather than MARB scale. Overall, numbers of animals in the MARB have decreased, but localized increases have occurred in several regions, which have had an impact on local water resources. The economic and environmental sustainability of AFOs hinges on reducing the nutrient imbalance at farm and watershed scales through carefully managed feeding strategies. The wider adoption of manure transportation that links producers with buyers will greatly enhance the sustainability of AFOs. The large-scale consolidation of AFOs, co-siting with biofuel production facilities (by-product grains used as animal feed), and increases in N fertilizer prices will likely create the economies of scale and alternative technologies for on-farm or localized manure use and management more feasible.

The success of nonprofit programs supported by watershed agricultural councils, industry, and state agencies should provide valuable demonstration models. If energy prices remain at current levels, bioenergy production from manures could provide an off-farm market for manures and reduce localized nutrient surpluses. Continuing educational efforts with farmers and the public regarding the importance and impact of conservation practices will be essential to reach environmental goals. Based on these findings, the Study Group offers the following recommendations.

- Strategies need to be implemented to encourage further development of alternative uses for manures, such as in composting, pelletizing, and granulation, and as a soil amendment in nutrient deficient areas of the MARB.
- Land-management planning and implementation of conservation practices should be designed to identify and avoid applications in critical loss areas, to use buffers or riparian zones, to manage grazing, to exclude stream banks, and to use subsurface injection with innovative applicators.
- Incentives to encourage on-farm and local bioenergy production from manure sources should be provided.

4.5.6 In-Field Nutrient Management

4.5.6.1 Fertilizer Sources

The principal fertilizer N sources (>90% of fertilizer N) used in the MARB are anhydrous ammonia, urea–ammonium nitrate solutions, and urea. Anhydrous ammonia use in several leading corn-producing states (IL, IN,IA, MN, NE, OH) has tended to decline in recent years, perhaps with the exception of consumption in Illinois and Indiana (Fig. 4.6) (Vroomen, H., Vice President, Economic Services, The Fertilizer Institute, 820 First St., NE, Washington, D.C., 2002, personal communication, 2007). The largest decline has been in Nebraska, where use of anhydrous ammonia N has declined about 40% since the mid-1980s.

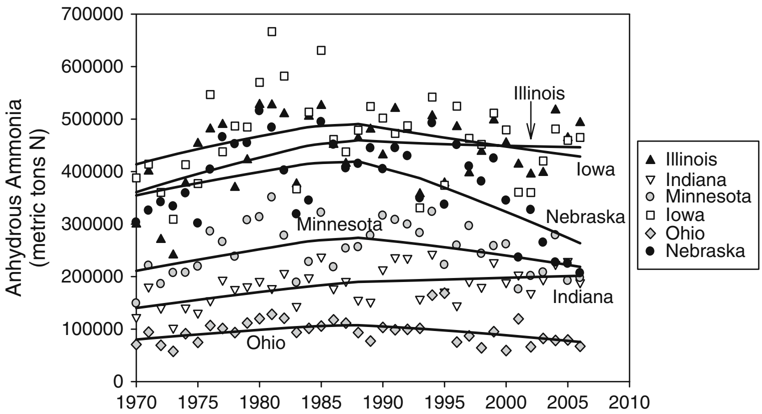

Fig. 4.6 Fertilizer N consumption as anhydrous ammonia in leading corn-producing states for years ending June 30

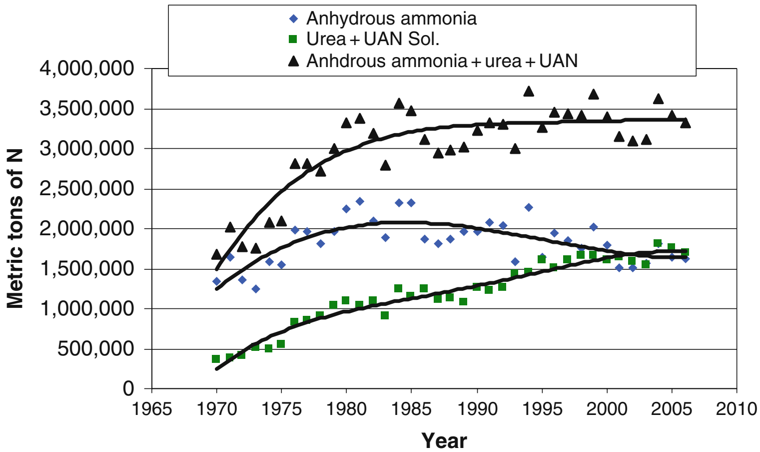

Fig. 4.7 Changes in the consumption of principal fertilizer N sources used in the six leading corn-producing states (IA, IL, IN, MN, NE, and OH) for years ending June 30

The combined N consumption of urea and urea–ammonium nitrate solution has increased and recently surpassed anhydrous ammonia tonnage in these six leading corn-producing states (Fig. 4.7). Although these data illustrate shifts in fertilizer N sources used, they do not allow conclusions about the portion of the annual anhydrous ammonia consumption that may be applied in the fall.

4.5.6.2 Fertilizer Use and Application Technology

The *Integrated Assessment* (CENR, 2000) concluded that "discharges of nitrogen from farms to streams and rivers could be reduced by implementing a wide variety of changes in management practices." These practices include switching from fall fertilizer N to spring N applications and applying nitrogen fertilizer and manure at not more than agronomically recommended rates. Application rate and timing are linked for N because the closer application is to the time of crop need, less N is lost to the atmosphere and water, and less N is needed. Research at five Management System Evaluation Area (MSEA) sites in the MARB (OH, IA, MN, MS, NE) reaffirmed BMPs for water quality, including soil nitrate tests, improved water management, and improved N timing and placement relative to crop needs (Power et al., 2000). Determining N sufficiency by monitoring for plant greenness and use of field or remote-sensing technologies followed by site-specific N applications hold promise to manage N more precisely.

Application timing. The risk of N loss with corn is greatest when fertilizer is applied some time before the period of rapid plant growth. Data on fall application are not directly available for the MARB and even seasonal data on fertilizer sales are not kept by all states in the MARB (Terry, 2006). Fertilizer sales records for Iowa (from July 2002 to June 2006) showed that 48% of N fertilizer was sold in the period from July to December and 52% from January to June. For anhydrous ammonia, the most common N form used and the primary form applied in the fall, 54% was sold in the period from July to December and 46% was sold from January to June. July to December sales of anhydrous ammonia accounted for 273,000 tons of actual N (data from http://www.agriculture.state.ia.us/fertilizerDistributionReport.htm). For Illinois, there has been an increase in fall N sales from the 1970s and 1980s to present, from about 25% to 40–50% (Fig. 4.8).

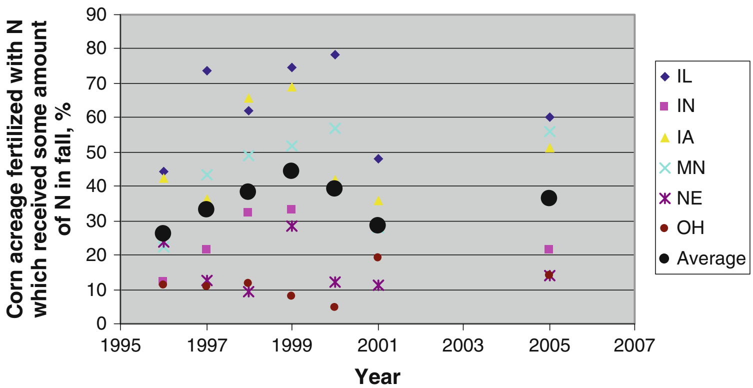

Fig. 4.8 Percentage of N-fertilized corn acreage that received some amount of N in the fall

Although it is not possible to correlate fall N application directly with fall fertilizer N sales, it is likely that a large fraction of the fall N sales represents N applied in the fall (Czapar et al., 2007). A portion of this fall N tonnage sold may also be stored at dealerships or in on-farm storage vessels for application the following spring. The USDA Agricultural Management Resource Survey (ARMS http://www.ers.usda.gov/Data/ARMS/app/Crop.aspx) data provide some insight into fall N applications, yet they are not sufficiently complete (i.e., key years are missing) to determine if the percentage of the acreage that receives some amount of fall N is increasing, decreasing, or remaining static (Fig. 4.8). The data do indicate that Minnesota, Iowa, and Illinois tend to fall apply some N on a larger fraction of their corn acres, compared to the other three states shown in Fig. 4.8. For three states, USDA ARMS data (Fig. 4.9) were used to calculate the total fraction of N applied

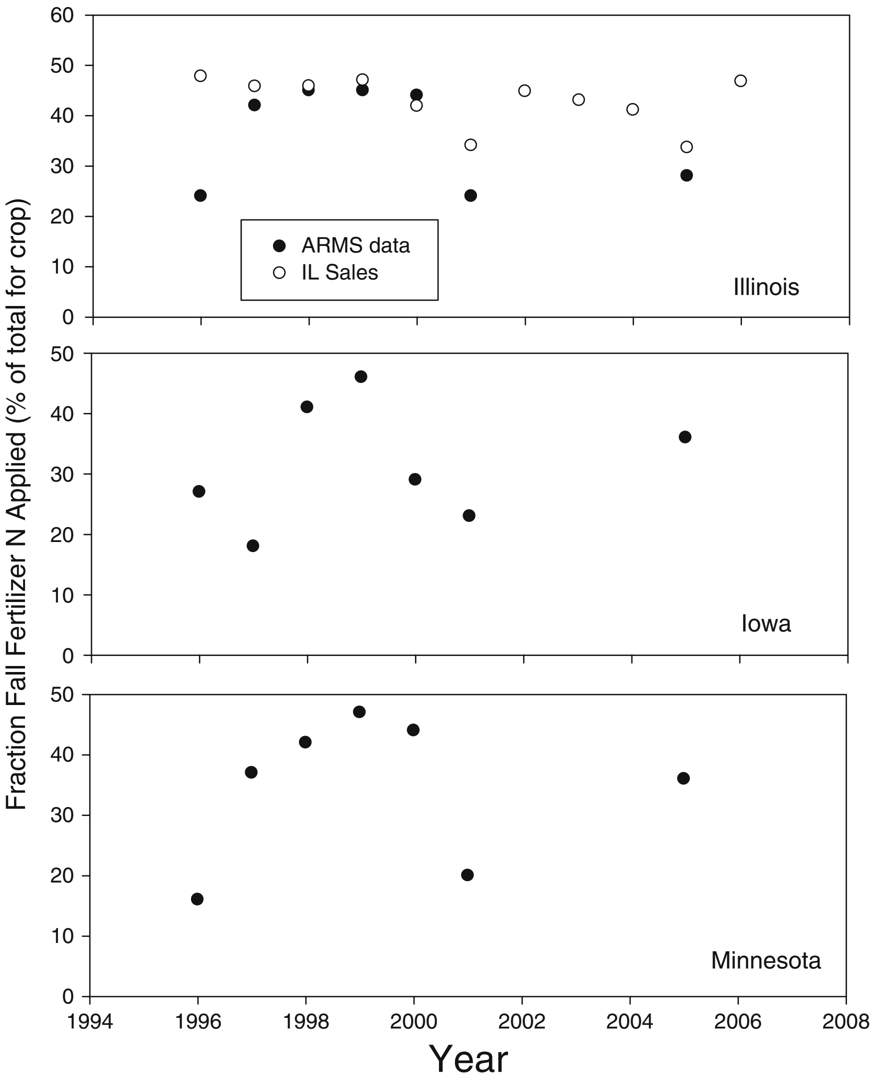

Fig. 4.9 USDA ARMS data for the three states with highest fall N application, showing total amount of fall-applied N for that crop. Also shown are Illinois sales data for the same period

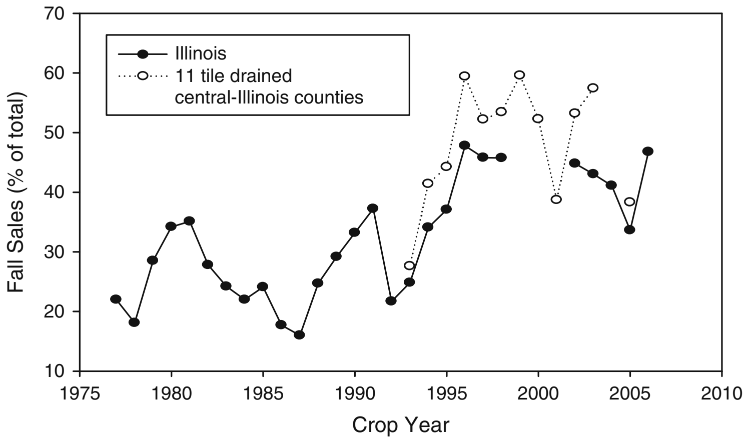

Fig. 4.10 Fraction of annual fertilizer N tonnage in Illinois sold in the fall

to corn in the fall, and IL sales were also compared (Fig. 4.10). As fewer producers in the Corn Belt farm the existing acreage, there has been greater pressure to complete fertilization in the fall, because of the numerous logistical challenges (labor demands, transportation and application equipment availability, weather uncertainty, and fertilizer supply and cost uncertainty) in the spring.

Randall and Sawyer (2005) contacted State Extension soil fertility specialists and State Fertilizer Associations to determine the fertilizer N amount that is applied in the fall. Based on these data, they estimated 25% (5.1 million hectares or 12.9 million acres) of the 20.5 million hectares (50.6 million acres) of corn in an 8-state area (IA, IL, IN, MI, MO, MN, OH, WI) received N in the fall. States with the largest amount of fall-applied N were Minnesota (1.85 million hectares or 4.56 million acres), Iowa (1.42 million hectares or 3.52 million acres), and Illinois (1.33 million hectares or 3.28 million acres). It is likely that tile-drained portions of these eight states have higher proportions of N applied in the fall either because of a greater dominance of corn/soybean agriculture or because regional soil temperatures are also cold enough to help minimize the conversion of ammonium-N to nitrate-N (nitrification) in the fall. Fall N application for corn as anhydrous ammonia is currently a recommended practice by virtually all Land Grant universities in the Corn Belt, where soil temperatures are consistently below 50°F at the 1.2–1.8 cm depth (0.47–0.72 in. or about ½–¾ in.), and the risk of environmental loss is not considered high or a pragmatic concern (Snyder et al., 2001). Additional guidance is usually provided in publications by Land Grant universities to maximize the benefits of fall N application and to help to minimize the risk of economical and environmental N losses (e.g., Bundy, 1998; Shapiro et al., 2003).

In a 2003 phone survey of Champaign Co., IL (a dominantly tile-drained area), 61% of the 352 respondents reported applying some N in the fall, and 49% of respondents applied all of their N in the fall (von Holle, 2005). Overall, the farmers who fall fertilized applied an average of 79% of their annual N needs before January 1, 2003. Data from 11 tile-drained central Illinois counties showed generally greater fall N fertilizer sales than the state as a whole (Illinois Department of Agriculture fertilizer tonnage reports). This difference was primarily due to the southern and nontile-drained portion of the state having winter soil temperatures that are too warm for fall application, where it is not recommended.

The effects of fall N application versus spring N application on nitrate transport in tile drainage depend on many factors, including soil temperatures, soil texture, precipitation, and drainage intensity. Randall and Sawyer (2005) reviewed the timing of N application and determined that spring application in Minnesota will typically result in 15% less nitrate-N loss than with fall application. In areas with warmer non-growing season temperatures (such as central Illinois) that are tile-drained, losses of fall-applied N may be greater. Watershed-scale studies of changing from fall to spring application (sidedressed) and changing the rate to account for more efficient use of spring-applied N showed at least a 30% reduction of nitrate concentrations in tile drain water (Jaynes et al., 2004). These studies indicate there is a great potential in some years for substantial reductions in N loss by applying N closer to when the crop can utilize it efficiently.

If these various estimates of N use and nitrate-N loss are combined, changing from fall to spring application may affect at least 25% of the corn acreage and reduce nitrate-N losses to streams from those acres by perhaps 10–30%. Split applications of N do not always result in increased N efficiency and reduced nitrate-N losses just because of improved N synchrony with crop uptake demands. The literature to support this practice indicates mixed results (Randall and Sawyer, 2005).

Nitrification inhibitors delay the conversion of ammonium to nitrate in soil. In Illinois, it is estimated that a nitrification inhibitor is added to about 50% of the fall-applied anhydrous ammonia (Czapar et al., 2007). Application of a nitrification inhibitor with anhydrous ammonia in the fall increased apparent recovery of N fertilizer in the corn grain from 38% without a nitrification inhibitor to 46% with an inhibitor, compared to 47% with spring application with no nitrification inhibitor in long-term research results in Minnesota (Randall et al., 2003; Randall and Sawyer, 2005). Ferguson et al. (2003) found that in Nebraska the benefits of nitrification inhibitors (either increased yield or reduced NO_3-N leaching) are strongly dependent on specific conditions and are most likely to be observed at suboptimal N rates (i.e., less than the economically optimum N rate [EONR; the point where the last increment of N returns a yield increase large enough to pay for the additional N]). They also reported that nitrification inhibitors can reduce crop yields with late sidedress N applications. It is well known that time of N application will largely govern any benefits from the use of nitrification inhibitors. Assuming increased N recovery by the crop translates to less nitrate leaching, nitrification inhibitors can potentially provide an economic benefit to farmers while reducing leaching.

Although the fertilizer N use trends indicate increased urea and urea–ammonium nitrate (UAN) solution use in the Corn Belt and lower anhydrous ammonia use (Fig. 4.7), there is a need for more research to document the benefits of split N applications of these two sources versus the more traditional fall anhydrous N applications. Use of urea and UAN solutions may provide greater flexibility in N management than has been experienced with anhydrous ammonia. Studies are underway to evaluate the crop and water quality effects associated with different N sources and time of application (e.g., see reports of work by Gyles Randall and others at the University of Minnesota: http://sroc.cfans.umn.edu/research/soils/index.html). In years when corn growth proceeds rapidly, timely sidedressing can be difficult, and delayed application can severely reduce yields (Randall, G., University of Minnesota, 2007, personal communication).

Application rate. Current N recommendations are usually applied across large geographic regions and may provide erroneous results for field-specific soil–crop–climate conditions (Gehl et al., 2005; Sawyer and Nafziger, 2005). Grouping soil types with similar drainage characteristics, rooting depth, and organic matter content is a feasible approach for determining more localized N recommendations and may result in more environmentally friendly N management (Oberle and Keeney, 1990). Remote sensing, geographic information systems, and variable application technologies offer an opportunity to develop and implement site-specific N recommendations, but the agronomic understanding of yield response to N on a site- and season-specific basis lags behind the technological innovations. There are instances, however, where considerable progress has been made in developing site-specific N recommendations (Raun et al., 2005).

Application of N near rates that provide the EONR usually results in drainage tile flow having nitrate-N concentrations in the range of 10–20 mg/L NO_3-N for soybean–corn rotations and 15–30 m/L NO_3-N for continuous corn (Sawyer and Randall, in press). Application of N above the EONR further increases NO_3-N losses and reduces net economic return. To the extent that N is being applied above the EONR, reductions in N loss through tile drains can be achieved with concurrent positive effects on net return (Sawyer and Randall, in press).

A review of the effects on N rates on corn–soybean systems in the upper MARB was conducted by Sawyer and Randall (2008) who found that in order to achieve a 30% reduction in tile drainage nitrate-N load, based on a study in Illinois, the N rate had to be reduced by 78 kg N/ha (70 lb N/ac) below the EONR, resulting in a large net economic loss ($67/ha or >$27/ac). These results illustrate an example of the risk of potentially large economic losses to farmers (and their communities) if they are asked to reduce N rates below their maximum net return or EONR (Sawyer and Randall, 2008). The potential environmental benefits of any N rate reductions are highly site specific and will also depend on how farmer's past N rates match their site-specific EONRs.

Economically optimum N rates are not the same across the Corn Belt states, and the same is true for other crops because of differences among soils, adapted crop varieties, climate, management, and many other factors that influence production and crop N requirements (Hong et al., 2006; Sawyer and Nafziger, 2005). Corn N needs vary widely both among and within fields (Lory and Scharf, 2003; Scharf et al., 2005). In some fields, in some areas of the MARB, where farmer's N rates have exceeded the EONR (especially where elevated N concentrations have been observed in water resources) there may be opportunities to reduce N rates for corn (Mamo et al., 2003) and other crops. Nitrogen application rate reductions must be economical for the farmer while also protecting water resources. Prior history of many management inputs including fertilizer N, manure, and tillage can affect crop N response and EONR interpretations. Farmers should carefully consider N rates and evaluate results over several years, in the same fields or plot areas. Rate reduction results obtained in 1 year can be highly affected by environmental conditions. For example, it is not uncommon to observe year-to-year variations in rain-fed corn yields ranging above 3.1–4.5 Mg/ha (50–90 bu/ac), and economic N rates associated with those yields to vary by more than 60–84 kg N/ha-year (54–75 lb N/ac-year) (Jaynes et al., 2001; Mamo et al., 2003; Sawyer and Randall, 2008).

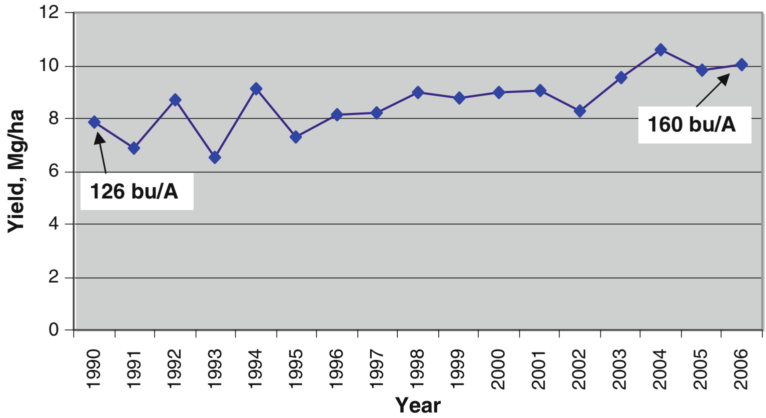

Fig. 4.11 Average corn yields in six leading corn-producing states (IA, IL, IN, MN, NE, and OH), 1990–2006 (Source: USDA National Agricultural Statistics Service)

As discussed in Section 3.2, higher crop yields (Fig. 4.11) have resulted in increased N removal in harvested grain, without increased N fertilization. Greater crop harvest N removal may have helped contribute to slight reductions in net N inputs in the entire MARB since about 2000, particularly in the Ohio and Upper Mississippi River subbasins (see Section 3.2), the two subbasins that also contribute the greatest annual and spring N flux to the NGOM. Increased crop yield trends, improved plant genetic selection, and pest control may also be contributing to the reduced nitrate-N transported to the NGOM since the mid-1990s, and the steady decline in total N delivered to the NGOM since the 1980s (see Section 3.1.1 and Fig. 3.8). Any reductions in N application rates could threaten attainment of high crop yields, which are vital to profitable production and which have contributed in some measure to the reductions in net N inputs and riverine N discharge mentioned above.

Challenges and complexities of determining the EONR in individual fields and farms prevent the ability to make any general conclusions regarding N rate reductions across the MARB that will achieve specific N load reductions to the NGOM.

Because of the complexity and dynamic nature of the N cycle, soil tests for N (nitrate, mineralizable N) have not met with much success in practical field applications (e.g., Scharf et al., 2006a). Some, like the Pre-Sidedress Nitrate Test (PSNT), have resulted in modest successes in N rate adjustments, particularly where there is a long history of manure applications and there has been a buildup of residual soil N (organic and inorganic). A new soil N test (ISNT) developed in Illinois offered promise of more reliably predicting mineralizeable soil N pools (Khan et al., 2001; Mulvaney et al., 2001); however, a recent report indicates the ISNT does not work well elsewhere (Barker et al., 2006a, 2006b; Laboski et al., 2006).

One of the key challenges in managing N in farm fields is to minimize unnecessary N applications in low-yielding years and to provide adequate N in high-yielding years to meet crop demands. Historically, it has been very difficult for even experts to predict residual soil N, recently applied fertilizer N, and mineralized N accessible by plants during a given growing season (e.g., Schlegel et al., 2005; Shehandeh et al., 2005). Furthermore, the inability to accurately predict the amount, intensity, or duration of rainfall in a given year makes it difficult to adjust N rates each year for a specific soil, crop variety/hybrid, tillage system, or cropping system.

4.5.6.3 Watershed-Scale Fertilizer Management

The first watershed-scale study of changing from fall to spring N application involves changes in both rate and timing (Jaynes et al., 2004). The Late Spring Nitrate Test (LSNT) is designed to help farmers add appropriate amounts of N in the spring instead of fall. Use of the LSNT for corn grown within a 400 hectares tile-drained watershed in Iowa resulted in at least a 30% reduction of nitrate-N concentrations in tile drain water. The LSNT involved changing timing, rate, and source of N fertilizer. Another Iowa study concluded that although watershed-scale implementation of LSNT had the potential to reduce nitrate loss through drainage water, it could also increase grower risk, especially when above-normal rainfall occurs shortly after the sidedress N is applied and N is lost to tile drainage or denitrification (Karlen et al., 2005). Development of affordable risk insurance or some other financial incentive by federal, state, or private agencies may be needed to stimulate adoption of the LSNT.

4.5.6.4 Controlled-Release Fertilizers

Controlled- and slow-release N fertilizers (CRN) are fairly commonly used in high-value applications, such as horticultural crop and turf production. Products include urea formaldehyde, isobutylidene diurea, sulfur-coated, and polymer-coated products. Use of CRN fertilizer is limited because of the high cost, with worldwide consumption less than 1% of all fertilizer N products. However, recent advances have brought some CRN products to an economical level for many agricultural crops. Controlled-release N fertilizers have the potential to significantly improve N use efficiency, maintain crop productivity, and minimize the potential for nitrate loss from fields (Blaylock, 2006).

4.5.6.5 Effects of N Management on Soil Resource Sustainability

It is well known that soil organic carbon (SOC) storage in Corn Belt Mollisols has been decreased by long-term cropping. For instance, in an Iowa study to determine the effects of cropping systems on SOC, there was 22–49% lower SOC than native prairie sampled in fencerows for all cropping systems that had been in place for 12–36 years (including continuous corn [CC]; corn soybean rotation [CS]; corn, corn oats, alfalfa; and corn oats alfalfa, alfalfa) (Russell et al., 2005). Current efforts to sequester carbon by restoring SOC and to obtain benefits of fertility and tilth associated with higher SOC in Mollisols should be considered in achieving nutrient load reductions from these crop production systems.

Nutrient management practices need to be assessed for their ability to enhance or maintain SOC content in addition to their impact on profit, yield, and water quality (Jaynes and Karlen, 2005). A careful review of the literature on this subject is warranted because of the potential that fertilizer management to achieve water quality improvements may lead to further soil quality degradation. Jaynes and Karlen (2005), based on Jaynes et al. (2001), find a partial N mass balance for three fertilizer N levels in a corn–soybean rotation on Mollisols in the Des Moines lobe region of Iowa. Tillage consisted of either moldboard or chisel plowing in the fall and use of a field cultivator for seedbed preparation and for weed control several times during the early growing season. The partial N mass balance shows that the 1X and 2X fertilizer N rates have a negative N mass balance, and the 3X rate has a positive mass balance. Although the 2X rate (134 kg N/ha or 120 lb N/ac on corn, no N applied to soybeans) was the economic optimum, the negative N mass balances may indicate a long-term decline in soil fertility. According to the authors, “the lower two N rates were thus effectively mining N from the SOM, which would result in a measurable decrease in SOM and a degradation of the soil resource over the long term.” Although all treatments had average nitrate-N concentrations above 10 mg/L nitrate-N, there were large and consistent differences among N loads in drain tile (Table 4.7). The 1X and 2X treatments achieved drain tile nitrate-N load reductions of 39 and 27%, respectively, compared to the 3X fertilizer N rate (201 kg N/ha or 179 lb N/ac).

The N mass balance approach to determining long-term changes in SOC or SOM presents numerous problems. First, there is no mechanism for lower fertilizer N applications to directly stimulate increased SOM mineralization. Any effect on SOC would be due to lower residue, particularly during the corn phase of the rotation and during soil tillage. Second, although a very high-quality study, the partial N mass balances shown are subject to different interpretations if only small errors exist. For instance, the total mass balance residual is less than 5% of the total fluxes measured and is 6–14% of the estimated N fixation. Therefore, small imprecision in estimated or measured values could lead to different interpretations.

A number of studies have made direct measurements of SOC over long-term studies of fertilizer rates. At least six relevant studies (three in IA and one each in KS, MN, NE) have been conducted on Mollisols in the Corn Belt. The general conclusion from these studies is that high fertilizer N rates on continuous corn will

Table 4.7 Partial N balance for 4-year rate study by Jaynes et al. (2001). The last two columns were added here and were not part of original table

	N inputs			N outputs						
Fertilizer rate	Total fertilizer applied kg N/ha	Total wet and dry deposition kg N/ha	Total fixed kg N/ha	Total grain removed kg N/ha	Total drainage loss kg N/ha	Total runoff kg N/ha	Change of residual mineral N kg N/ha	N balance residual %	(Residual/ fixed)*100 %	(Residual/ total flux)*100 %
1×	144	43	395	522	119	0	6	–55	–14	–4.4
2×	289	43	397	590	142	0	13	–26	–6.5	–1.8
3×	414	43	394	606	195	0	–7	47	12	2.8

lead to SOC increases and that suboptimal N rates lead to SOC depletion. There is no direct evidence for an effect of lower nonzero fertilizer rates near the economic optimum, leading to decreases in SOC from these studies.

Russell et al. (2005) analyzed studies of two Iowa sites (Kanawha and Nashua) for the impact on SOC of four N fertilization rates (0, 90, 180, and 270 kg N/ha-year or 0, 80, 161, and 241 lb N/ac-year) and four cropping systems (continuous corn [CC], corn soybean [CS]; corn–corn–oat–alfalfa [CCOA], and corn–oat–alfalfa–alfalfa [COAA]). One study had been ongoing for 23 years and the other for 48 years at the time of sampling of SOC in 2002. The only difference related to fertilizer rate was for the 23-year experiment (the Nashua site). In this experiment, the 270 kg N/ha-year (241 lb N/ac-year) for CC had higher SOC for only the 0–15 cm (0–5.9 in) depth. There were no differences among the 0, 90, and 180 kg N/ha-year rates for CC at the Nashua site for any depths. There were also no differences for the 0–100 cm (0–39 in) soil for any N rates used for CC, including the highest rate of 270 kg N/ha-year (241 lb N/ac-year). There were no other significant fertilizer N rate effects found in the study (Russell et al., 2005).

An earlier Iowa study that included the Nashua and Kanawha sites and a third site (Sutherland) reached similar conclusions as those of Russell et al. (2005). In that study, Robinson et al. (1996) found that N fertilizer rate on corn (0–180 kg N/ha-year or 0–161 lb N/ac-year) was not significant in determining SOC but only whether fertilization occurred. In both studies (Robinson et al., 1996; Russell et al., 2005), the cropping systems with alfalfa [termed meadow in Robinson et al. (1996)] had the highest SOC. Corn silage treatments and no fertilizer treatments had the lowest SOC (Robinson et al., 1996). A third Iowa study did not compare SOC under different fertilizer rates but did show that high fertilizer N (206 kg N/ha-year or 184 lb N/ac-year) resulted in increases in SOC over 15 years with continuous corn (Karlen et al., 1998a). The general conclusion from the Iowa studies is that, for either CC or CS systems, fertilizer rate has little or no effect in the 90–180 kg N/ha-year (80–161 lb N/ac-year) range. Given that the average N fertilizer application to corn in Iowa was 158 kg N /ha (141 lb N/ac) in 2005 (USDA ERS: http://www.ers.usda.gov/Data/ARMS/app/CropResponse.aspx) and the economic optimum rate ranged between 67 and 172 kg N/ha or about 60 and 154 lb N/ac (approximate mean of 137 kg N/ha or 122 lb N/ac) during 1996 and 1998 in the Iowa study by Jaynes et al. (2001), it seems unlikely that these rates would lead to a depletion of SOC due to a N rate effect. Corn yields with the moderate N rates in the Jaynes et al. (2001) study ranged around 10 Mg/ha (159 bu/ac), and the Iowa state average corn yield in 2005 was about 10.9 Mg/ha (173 bu/ac).

Results from other studies in the Corn Belt are mixed and have found no consistent effect of N rate on SOC. In Kansas, Omay et al. (1997) found no effect of either 224 or 252 kg N/ha (200 or 225 lb N/ac) versus no N for over 10 years of CC or CS. A small significant difference in SON (less than 5% decrease) was found on one soil for the zero N treatment. Increased residue inputs were attributed to N fertilization and inclusion of soybean in the rotation reduced SOC and soil organic N. In contrast, CC receiving 200 kg N/ha-year (179 lb N/ac-year) for 13 years had higher SOC than in the zero N treatment on a Minnesota Mollisol (Clapp et al., 2000). In an 18-year experiment in Nebraska, N rate (0, 90, 180 kg N/ha-year or 0, 80, 161 lb N/ac-year) had an effect on SOC in the 0–7.5 cm (0–2.9 in) soil after 8 years but had no effect after 18 years, presumably due to tillage differences (Varvel, 2006).

Recent work in Nebraska on an irrigated Mollisol compared long-term (initiated in 1999) continuous corn and corn–soybean rotations under recommended and intensive management and found that SOC was increased under recommended and intensive management of CC but not in the CS systems (Adviento-Borbe et al., 2007; Dobermann et al., 2007). These scientists also reported that greenhouse gas (GHG) emissions from agricultural systems can be kept low when management is optimized toward better exploitation of the yield potential. To accomplish SOC increases while keeping GHG emissions low, Dobermann et al. (2007) reported the following required factors: (1) choosing the right combination of adapted varieties or hybrids, planting date, and plant population to maximize crop biomass production; (2) tactical water and N management, including frequent N applications to achieve high N use efficiency and minimized N_2O emissions; and (3) a deep tillage (noninverting) and residue management approach that favors a buildup of SOC as a result of large amounts of crop residues returned to the soil.

If a fertilizer effect on SOC exists, it is more likely to occur under CC than CS because increased fertilizer generally leads to increased corn production. It is logical to assume that increased corn production (including grain, stover, and roots) should lead to increased SOC. In general in the published studies, this relationship does not hold, although applying zero N fertilizer generally leads to less SOC over time than high fertilizer N rates. In summary, although it is beyond the scope of the Study Group to review all the research relevant to changing SOC in Corn Belt soils, it is clear that inclusion of alfalfa in a rotation is very effective at building SOC. The effects of tillage are not clear. Based on the existing literature, there is evidence that changes in fertilizer rates within the range of those optimum for corn production are unlikely to lead to long-term SOC and SON declines. Although it is possible to build SOC under CC with relatively high fertilizer additions [e.g., 201–299 kg N/ha-year or 179–267 lb N/ac-year (Adviento-Borbe et al., 2007; Dobermann et al., 2007) and 206 kg N/ha-year (184 lb N/ac-year) Karlen et al., 1998b], care must be taken to ensure that these fertilizer additions are sustainable economically and that they do not harm water quality. From a global C balance perspective, it is also worth noting that there is a C emissions cost of producing N fertilizer that would need to be taken into account when doing C mass balances for higher fertilizer N rates on corn. However, if high-yield production is achieved, with good N use efficiency, these fertilizer C emissions may be offset (Adviento-Borbe et al., 2007). More research on the net effects of N fertilizer rates on SOC and GHG emissions is needed.

4.5.6.6 Precision Agriculture Management Tools for Nitrogen

Global positioning system (GPS) and geographic information system (GIS) technologies are becoming more widely adopted by farmers and show promise for developing management zones in fields that could target application rates for low- versus high-yielding areas (Schlegel et al., 2005) and reduce N applications in areas of the field most prone to N losses (Chua et al., 2003). Field-transect apparent electrical conductivity (ECa) or electromagnetic induction measurements can help define management zones, based on surrogate detection of soil texture differences (Davis et al., 1997; Kitchen et al., 1999). Reductions in N application rates for corn range from 6 to 46% when using site-specific management zone approaches as opposed to a uniform rate of N application (Koch et al., 2004). Dividing fields into a few management zones might reduce N loss, but because of within-field variability, more spatially intensive N management might provide greater economic and environmental benefits (Hong et al., 2006; Scharf et al., 2005).

Basing N applications on past yields has not proven to be an effective approach to variable-rate fertilization of N (Murdock et al., 2002; Scharf et al., 2006b). In-season crop N-sensing research (chlorophyll meter, remotely sensed multispectral color images, on-the-go and handheld optical reflectance sensors) (Scharf et al., 2006a), using reference "N-rich" or calibration strips or plots in targeted areas within fields (Raun et al., 2005), has shown the potential benefits of these newer technologies in providing in-season guidance to farmers and crop advisers for improved N nutrition management. This "N-rich" calibration approach appears to have been more successful with winter wheat than for corn, to date. Chlorophyll meters and remotely sensed crop reflectance have been used as an index for plant N status, and N-fertilizer use efficiency improved when these techniques were used (Osborne et al., 2002; Varvel et al., 1997). Crop N-sensing technologies present opportunities to reduce and better time fertilizer N applications; however, there have been few direct assessments of impacts of these approaches on residual soil N and nitrate losses. Further verification of the performance of these techniques is needed in order for implementation by farmers to be more widespread.

When technology costs are considered, economic returns to farmers are often inadequate to justify adoption of variable-rate N management. Frequently, the costs of spatial N management technologies exceed the cost of the fertilizer N saved, which are dependent on fertilizer prices. As a consequence, adoption of these technologies has proceeded at a slower rate than anticipated, partly because of high technology and equipment costs and spatially variable economic returns. Economics research suggests a number of reasons for this low slow adoption, including high fixed costs of adoption and uncertainty in returns. These factors suggest that incentives to encourage adoption may need to cover option values and that revenue insurance programs to address the risk may be appropriate instruments (Khanna, 2001; Khanna et al., 2000; Isik and Khanna, 2002, 2003).

Incentives have been used in Missouri in cost-sharing some of the expenses of precision technologies within the USDA EQIP program (Agronomy Technical Note MO-35, September 2006). Cost share in this Missouri USDA-NRCS Code 590 nutrient management program provides a farmer \$49/ha (\$20/ac) per year for a 3-year contract, with the full \$148/ha (\$60/ac) provided at the end of the first year. Farmers in this Missouri EQIP precision N-sensing program are advised to follow guidance for N-sensing interpretation based on work by Scharf et al. (2006a, 2006b).

4.5.6.7 Precision Agriculture Management Tools for Phosphorus

Spatial variability in soil test phosphorus (P) levels can be large, with levels often ranging from very low to very high (agronomic interpretation) in the same field (Bermudez and Mallarino, 2007; McGraw and Hemb, 1995; Reetz et al., 2001; Wittry and Mallarino, 2004). This variability can also be large in fertilized, manured, and grazed pastures (Mallarino and Schepers, 2005; Snyder and Leep, 2007). With the advent of commercially available GIS and GPS technologies in the early 1990s, crop advisers and farmers began to more precisely define the spatial variability of soil fertility levels, including soil test P (Fig. 4.12). In recent years, zone or grid (e.g., 0.25–1 hectare or 6–2.5 acres) sampling has been used to better define management units to receive different P application rates (Reetz et al., 2001), as opposed to the formerly recommended practice of whole-field composite sampling (e.g., Thom and Sabbe, 1994). In spite of considerable research effort, no widely accepted standard for soil sampling fields for precision or site-specific management has been established (Mallarino and Schepers, 2005), because soils are naturally heterogeneous and their spatial variability occurs at many scales. Recent soil sampling summary results for more than 3.3 million soil samples in North America from both public and private soil testing laboratories also showed wide variability in soil test P levels within and among states in the United States (PPI/PPIC/FAR, 2005). Snyder (2006) summarized the soil test results for the 20 major MARB states (over 2.1 million samples) and reported (1) 40% of the states have experienced a decline in soil

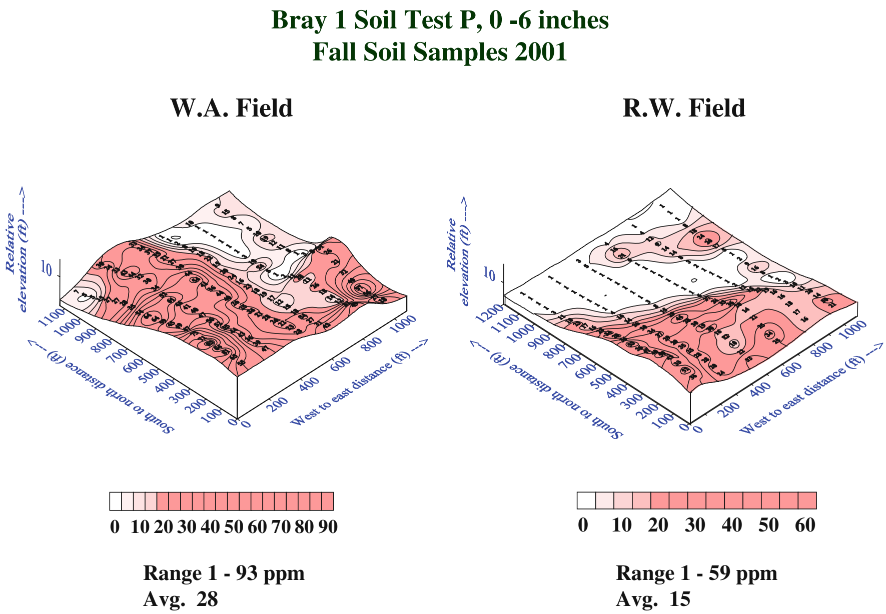

Fig. 4.12 Variability in soil test P levels in typical farmer fields in Minnesota (2007 personal communication with Dr. Gary Malzer, University of Minnesota)

test P since 2001, and (2) 78% of the samples tested below 50 mg/kg (ppm) Bray 1 equivalent-extractable P and 94% tested 100 ppm or below. In fact, crop harvest removal of P exceeds fertilizer plus recoverable manure P in 11 of the 20 states (PPI/PPIC/FAR, 2002). These data are in agreement with the trends in net anthropic P input in the MARB, discussed in Section 3.2 of this book.

Early season detection of corn P deficiency may be possible with remote sensing, but detection of deficiencies later in the season, which correlate better with crop yield, has not been successful (Osborne et al., 2002). At this time, remote sensing or on-the-go sensing of plant P status does not appear to be as commercially viable as plant N sensing.

Variable-rate fertilization can result in better P fertilizer management. For example, Bermudez and Mallarino (2007) found that variable-rate technology applied 12–41% less fertilizer and reduced soil test variability on farmers' fields in Iowa, compared with the traditional uniform rate fertilization method. Perhaps one of the most important aspects of intensive soil sampling and variable-rate P application technologies is the capability to apply P fertilizer where it is needed while minimizing or reducing P applications in field areas which have elevated soil test P. In Iowa, variable-rate P application helped decrease soil test P in field areas with high soil test P, when applying manure (Fig. 4.13) or fertilizer (Fig. 4.14). As of yet, however, variable-rate or precision P fertilization has been shown to have little economic benefit in the major corn and soybean producing states compared to uniform applications (Lambert et al., 2006; Mallarino and Schepers, 2005). Further, there are ongoing efforts to update soil test P crop response calibrations and fertilizer recommendations to optimize P fertilization (Beegle, 2005).

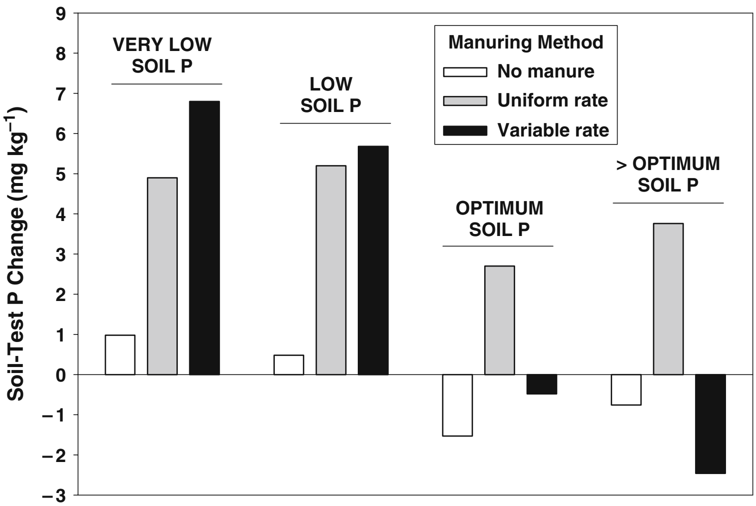

Fig. 4.13 Effect of variable-rate versus uniform-rate application of liquid swine manure on changes in soil test phosphorus in Iowa fields [2007 personal communication with Dr. Antonio Mallarino, Iowa State University and Wittry and Mallarino (2002)]

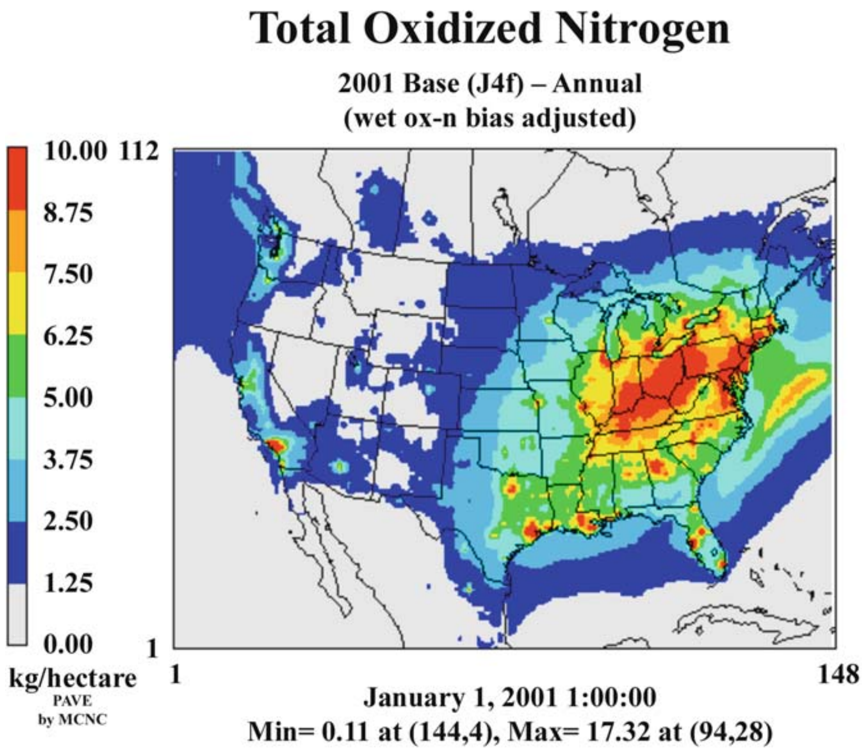

Fig. 4.14 Effect of variable-rate versus uniform-rate application of fertilizer P on soil test P in multiple Iowa fields across multiple years

Numerous studies have shown a strong relationship between soil test P levels and the concentration of dissolved P in runoff (Andraski and Bundy, 2003; Pote et al., 1999; Sharpley et al., 2006a, 2006ba) and tile drainage (Heckrath et al., 1995). Recent work by Gentry et al. (2007) showed that tile drainage P losses in Illinois can exceed 1 kg P/ha-year (0.9 lb P/ac-year), with much of the loss occurring during a few peak storm events in the spring. However, annual manure or fertilizer P applications can control the concentration of total and dissolved P in surface runoff (Pierson et al., 2001; Sharpley et al., 2001).

Soil test P thresholds alone cannot define the potential or risk of P losses from agricultural fields. Slope, hydrologic characteristics, tillage, P rate, and time after P application before a runoff producing rainfall, and other factors also affect the risk of P loss (Sharpley et al., 2006a). To address all factors influencing P loss from agricultural fields, an environmental risk assessment tool (the P Index) was proposed by Lemunyon and Gilbert (1993), which has been regionally modified and adopted by 49 of 50 states in the United States to identify and delineate the risk for agricultural P loss for use in the development of Comprehensive Nutrient Management Plans

(Sharpley et al., 2003). Use of P Indices has also been encouraged by industry, in recognition of the spatial variability in soil test P levels within fields, and the spatial variation in source and transport factors (Snyder et al., 1999). "Variable rate P application can be practically implemented on the basis of P index ratings for field zones, not just based on soil test P" (Wortmann et al., 2005). Variable-rate fertilizer P application is becoming more common in Nebraska, Iowa, Missouri, Kansas, and other states, and some custom applicators are beginning to apply manure at variable rates.

4.5.6.8 Nutrient Management Planning Strategies

A survey of 127 farms (90% of all farms) in two northeastern Wisconsin watersheds offers some insight into how successful nutrient management has been in reducing nutrient applications (Shepard, 2005). Farmers with a nutrient management plan (53% of farms) applied less N and P (139 kg N/ha and 31 kg P/ha or 124 lb N/ac and 28 lb P/ac) than farms without a plan (188 kg N/ha and 44 kg P/ha or 168 lb N/ac and 39 lb P/ac), but only half the farmers credited on-farm manure N, and only 75% fully implemented their plans on most of their acres.

For nutrient management planning to decrease nutrient loss, technical and financial assistance programs need to focus on plan implementation and maintenance in the MARB rather than on targeting the number of plans written in a given period. Despite programs subsidizing plan writing, a critical limitation is the lack of certified plan writers to meet the demand and deadlines. Further, there needs to be an effective mechanism to ensure plan adoption and regular updating of plans. Efforts are underway in the Heartland states of the MARB (IA, KS, MO, NE) to develop nutrient management plan assessment protocols. This aims to identify key factors that limit plan implementation so that practical solutions can be developed. One option is preparation of a simplified plan that farmers can quickly refer to. Also, documenting nutrient management plan implementation is being rewarded with financial credits in New York drinking water supply watersheds (Watershed Agricultural Council, 2004). These credits can be used to purchase or upgrade equipment that would need to be used to implement the plan, such as manure spreaders and injectors.

An assessment is needed of the socioeconomic barriers to successful adoption of nutrient management planning strategies in the MARB as well as the N and P loss reductions achievable. Such an assessment has been done in a drinking water supply watershed for New York City that claims a 93% participation in volunteer conservation programs (Watershed Agricultural Council, 2004). A survey of CREP participants showed that they were generally older and more likely to obtain information from extension agents, consultants, and watershed council personnel than nonparticipants, but there was no difference in educational level or farming status (full or part time) (James, 2005). Overall, negative attitudes toward voluntary adoption of BMPs were a result of the loss of productive land and loss of being able to decide independently what to do on their own land. These survey results illustrate the difficulties in gaining adoption of nutrient management BMPs by farmers in any watershed, transferring new BMP technology, and addressing the socioeconomic pressures faced.

Key Findings and Recommendations

Reductions in N losses and residual soil NO_3-N are possible with attention to improved infield N management. It may be possible to reduce N rates and alter N timing in some portions of the MARB. Such rate reductions may be accomplished through implementation of refined management, but they must be economical for farmers and care must be taken to protect soil resource sustainability. Crop N sensing and variable-rate N management implementation, using management zone approaches may prove useful in attainment of economic optimum N rates in individual fields, which may also help reduce N losses. Higher fertilizer, fuel, and machinery costs have stimulated increased interests in some newer N management technologies, as well as other means to improve fertilizer N effectiveness and efficiency; however, use of site-specific or precision technologies has not yet proven financially rewarding to many farmers, due to the high cost of sampling, ground-truthing, and application technology. Based on these findings, the Study Group offers the following recommendations.

- Because of the importance of both N and P to Gulf hypoxia and as various cropping systems can have different positive and negative effects on N and P export reduction, remedial strategies must be directed at system-wide nutrient management rather than either N or P applications alone. Future research to evaluate the effects of different nutrient management impacts on crop production should include measures of water and air quality effects.
- There is a lack of consistent year-to-year USDA nutrient management survey data, which hinders any broad nutrient use and management evaluation and interpretations. These data will become more important in monitoring and understanding changes in nutrient management practices as biofuel markets expand. Consistent year-to-year data collection on nutrient management of major crops and emerging energy crops is recommended.
- Cost-share incentives like the USDA payment support for crop N-sensing and precision N management in Missouri, intensive educational programs (e.g., on-farm demonstrations), and/or other means should be explored to encourage the agricultural community to improve nutrient use efficiency and effectiveness with all nutrient sources (i.e., fertilizer, manure, biosolids, composts, by-products, etc.). Such programs may be especially helpful in corn systems in the upper Mississippi and Ohio River

subbasins, which have been identified as major contributors of spring nitrate-N flux to the NGOM.

- Although the economic and water quality impacts of controlled-release fertilizers in commercial field crop systems have not been fully proven, their beneficial use should be explored through additional research and demonstrations at field and watershed scales. Programs to stimulate greater adoption of locally proven technologies like urease and nitrification inhibitors (and controlled-release fertilizers, once proven economically and environmentally effective) to enhance crop nitrogen recovery and use efficiency should be considered as the shift toward greater urea and urea–ammonium nitrate N use continues.
- Watershed-scale evaluations of split applications of N in the spring for corn should be conducted to determine watershed-scale benefits of this N management approach compared to the more traditional application of anhydrous ammonia in the fall, especially in the upper Mississippi and Ohio River subbasins.
- More research on the net effects of N fertilizer rates on soil organic carbon (SOC) and greenhouse gas (GHG) emissions is needed.
- Crop and animal production systems are essential to the economic viability of agriculture in the MARB. Thus, an infrastructural assessment of how animal production can co-exist with grain and forage production is needed. Long-term strategies should be explored whereby more effective crop and animal production systems remedy or avoid excessive N and P loading to water and air resources.
- Cost–benefit ratios vary among farmers with, for example, labor availability, farm organization, and financial situation. However, past experience shows that adoption of conservation practices is not solely dependent on cost-effectiveness. Thus, there needs to be consideration of the socioeconomic barriers to, and impacts of, adoptions of nutrient management planning strategies in the MARB. New approaches should be investigated to overcome socioeconomic barriers, including incentive programs.

4.5.7 Effective Actions for Other Nonpoint Sources

4.5.7.1 Atmospheric Deposition

This section reviews actions for reducing NOx emissions that contribute to atmospheric deposition of nitrogen. Atmospheric deposition of oxidized nitrogen compounds released during fossil fuel combustion contributes an estimated 30% of the entire inputs of new nitrogen for the United States as a whole, (Howarth et al., 2002). As discussed earlier in Section 3.2, atmosphere deposition of oxidized nitrogen is less important in the MARB but still accounts for an estimated 8% of nitrogen contributions to the upper MARB and 16% of the nitrogen inputs to the Ohio River basin. NOx emissions to the atmosphere in the United States could be virtually eliminated at reasonable cost using currently available technologies (Moomaw, 2002; Howarth et al., 2005). In addition to potential benefits concerning Gulf hypoxia, reducing NOx emissions in the MARB can contribute to improved local air and water quality and can reduce atmospheric transport of nitrogen to the northeastern states, where atmospheric deposition is an even more significant problem.

In addition to deposition of oxidized nitrogen, there is significant deposition of ammonia and ammonium (NH*x*) in some regions of the MARB. These are not considered in the mass balance approach for nitrogen in Section 3.2 because the NH*x* originates largely from volatilization from animal wastes and other agricultural sources and so does not represent new nitrogen inputs to the basin, but rather a recycling of nitrogen within the basin (Howarth et al., 1996). Nonetheless, high rates of volatilization followed by conversion to ammonium nitrate or sulfate can lead to significant long-distance transport and contribute to reactive N distribution in other sensitive areas. Furthermore, high rates of NH*x* deposition in the basin can result in increased leakage of nitrogen to downstream aquatic ecosystems. In Iowa, Minnesota, and Wisconsin, NH*x* deposition exceeds NO*y* deposition, and averages over 7.5 kg N/ha-year (6.7 lb N/ac-year) in Iowa (results from CMAQ model, Robin Dennis, NOAA, unpubl.)

Mobile sources account for approximately 55% of NO*x* emissions to the atmosphere on a national level (Melillo and Cowling, 2002). While automobiles have been subject to fairly strict NO*x* standards in recent years, emissions from light trucks have not historically been as strict. Tightening regulations on light trucks represents an opportunity for significant reduction in NO*x* emissions, as approximately half of new vehicle sales in recent years have been light duty trucks (Moomaw, 2002). Heavy diesel trucks, buses, and trains have accounted for a growing fraction of NO*x* emissions because of strict NO*x* standards on automobiles and the absence of similarly strict controls on heavy diesel vehicles.

Stationary sources account for approximately 45% of NO*x* emissions, with electric generating facilities accounting for roughly half of all stationary source emissions, and industrial fuel combustion account for slightly less than one-third. The remainder of stationary source NO*x* emissions is from nonfuel industrial processes (12%) and from commercial, institutional, or residential fuel combustion (8%) (USEPA, 2006a).

Stringent new source performance standards have greatly reduced emissions from new electric generating facilities. Low-emission, combined-cycle gas turbines account for most new electric generating capacity in recent years (Bradley and Jones, 2002). Unfortunately, some existing policies provide incentives that discourage more widespread adoption of new, cleaner technologies. For example, under the Clean Air Act, high NO*x* emissions by older, coal-fired power plants are "grandfathered," and therefore not subject to the stringent emission standards of new generating capacity. As a consequence, electric utilities have the incentive to keep older coal plants running far beyond what would otherwise be their economic lifespan (e.g., Ackerman et al., 1999; Maloney and Brady, 1988; Nelson et al., 1993). As a result, while 90% of new electric generating capacity is produced with gas turbines, coal still produces 55% of the electricity in the United States (Moomaw, 2002). And it was estimated that, in 1998, coal-fired power plants were responsible for nearly 90% NO*x* emissions from electric power generation (USEPA, 2000b, 2006a). About a quarter of the coal-fired electric generating capacity in 1996 was constructed prior to 1965, and almost one-half was constructed prior to 1975 (Ackerman et al., 1999).

Considerable reductions in NO*x* emissions can be achieved with existing commercial technologies by replacing outdated coal-fired capacity with modern gas-fired combined-cycle power plants (Howarth et al., 2005). Existing coal plants can also be retrofitted with new control technologies, such as low-NO*x* burners (Ackerman et al., 1999; Bradley and Jones, 2002). Other promising technologies for reduction emissions from coal-fired power plants include fluidized-bed boilers (Cogeneration Technologies, 2006) and gasified coal combined-cycle power plants (U.S. DOE, 2006).

For the most part, NO*x* emissions in the United States are regulated because of concerns over formation of smog and ozone and seldom because of water quality concerns (Melillo and Cowling, 2002; Moomaw, 2002). Since smog and ozone pollution occur mostly in summer months, regulation of NO*x* emissions from stationary sources has often focused on summertime only regulation (Howarth et al., 2005). Since the largest cost of controlling NO*x* from power plants is the capital cost of building scubber systems, the additional cost of requiring year-round NO*x* control from power plants is small compared to that for summertime only controls. Thus, year-round operation of existing control technologies represents a cost-effective approach for reducing NO*x* emissions. Some local and state governments, such as New York State, have recently moved toward year-round regulation of NO*x* because of concern over coastal nitrogen pollution (Ron Entringer, NY State DEC, personnel communication).

4.5.7.2 Residential and Urban Sources

Urban and suburban runoff comes from a variety of sources, including impervious surfaces like roads, rooftops, and parking lots, as well as pervious surfaces like lawns. Urban and suburban runoff can be important sources of pollutants, especially for local water quality effects. For example, the *National Water Quality Inventory: 2000 Report to Congress* concluded that urban runoff is a major source of water quality impairment in surface waters (USEPA, 2002). A variety of actions can be used to control nonpoint urban sources, including both structural and nonstructural practices (e.g., USEPA, 2005).

Although controlling urban nonpoint sources can provide significant benefits from improvements to local water quality, these nonpoint sources are not significant determinants of hypoxia in the Gulf of Mexico, both because concentrations tend to be lower than those from agricultural sources and because the urban land comprises less than 1% of the Mississippi River basin (e.g., Mitsch et al., 1999). Thus, although actions to reduce urban nonpoint sources may be justified, these control actions will not likely contribute significantly to reductions in the size of the Gulf of Mexico hypoxic zone. Since control of urban nonpoint sources will not have an important role in reducing hypoxia, we do not focus on actions to reduce urban nonpoint sources of nutrients in this book.

Key Findings and Recommendations

Atmospheric deposition is a small but significant (8% in Upper Mississippi and 16% in Ohio River subbasins) contribution to N inputs in the Mississippi River basin. Opportunities exist to lower NO*x* emissions in a number of ways, but it is not likely that hypoxia will drive most of these regulatory decisions. Rather, hypoxia reduction and other water quality benefits should be incorporated in a number of regulatory decisions regarding air pollution. Based on these findings, the Study Group offers the following recommendations.

- Water quality benefits and effects on hypoxia should be incorporated into decisions involving retirement or retrofitting of old coal-fired power plants; NO*x* controls, such as the extension of the current summertime NO*x* standards to a year-round requirement; and emissions standards; and mileage requirements for sport utility vehicles, heavy trucks, and buses.

4.5.8 Most Effective Actions for Industrial and Municipal Sources

Sewage treatment plants and industrial dischargers represent a more significant source of N and P in the MARB than was originally identified in the *Integrated Assessment*. Although most point sources in the MARB do not have permits that require removal of N or P from discharged effluent, as local water quality standards for these nutrients have not yet been developed, states are charged with developing water quality criteria for achieving and maintaining designated beneficial uses of surface waters, including those waters that receive sewage treatment plant effluent. However, the process by which these criteria are translated into quantitative and enforceable nutrient limits from regulated point sources remains unclear.

Based on data from the recent MART (2006b) report, the Study Group has estimated that permitted point source discharges represented approximately 22 and 34% of the average annual total N and total P flux to the Gulf, respectively, for the 2001–2005 water years (for a detailed discussion see Appendix D). These point sources represent a significant opportunity to reduce N and P loadings that should be fully evaluated in the context of other potential management changes in the MARB.

Encouraging behavioral changes of nondomestic sewer users as well as increasing capital investments in sewage treatment and industrial treatment plant upgrades have proven to be effective approaches to managing nutrient discharges in other areas of the United States (USEPA, 2004b, 2003a; Chesapeake Bay Commission, 2004). The use of Biological Nutrient Removal and Enhanced Nutrient Removal technologies for N and P removal is being implemented to reduce N and P concentrations in sewage treatment plant effluent discharge by 50–80% (Maryland Department of Environment, 2005; USEPA, 2004b). Sewage treatment plant upgrades designed to remove phosphorus typically include enhanced chemical precipitation applied alone or in combination with biological phosphorus treatment and membrane filtration. These types of sewage treatment plant unit operations, which can achieve effluent discharge phosphorus concentrations as low as 0.1 mg/L total phosphorus or less, now constitute the BMP for phosphorus removal at sewage treatment plants. Removing P to a 0.1 mg P/L limit is most commonly implemented where there is a market for water recycling, such as in communities located in the desert Southwest, and the increased cost can be justified. In locations where there is no market for recycled water, higher limits for P (for example, 0.3 or < 1.0 mg P/L) will be more cost-effective.

The Study Group presents an example calculation to demonstrate the magnitude of reduction possible in riverine total N and P fluxes to the NGOM if technology for N and P removal from sewage effluent were implemented for large sewage treatment plants (0.5 million gallons per day and above) across the MARB. Based on the Study Group's adjustment to the MART report's estimates of N and P effluent from sewage treatment plants (MART, 2006b), the Study Group has calculated that upgrades for large sewage treatment plants in the MARB to achieve total N concentration limits of 3 mg/L could create reductions in N flux from sewage treatment plants from 192,000 metric tonne N/year (212,000 ton N/year) to 70,000 metric tonne N/year (77,000 ton N/year), about a 64% reduction in annual N flux from sewage treatment plants. This translates into a reduction of total annual N flux to the Gulf by about 10% and the total spring N flux by about 6%. Upgrading to achieve P concentrations of 0.3 mg/L would create reductions in P fluxes from sewage treatment plants from 41,000 metric tonne P/year (45,000 ton P/year) to 10,500 metric tonne P/year (11,600 ton P/year) or about a 75% reduction in annual flux from sewage treatment plants to the MARB. These reductions, in turn, would translate into a decrease in the total annual P flux to the Gulf by about 20% and the total spring P flux by about 15%. It is important to recognize that these estimates assume that the changes in biosolids quality and production rates resulting from the capital improvements to the sewage treatment plant do not adversely impact nutrient management procedures implemented at biosolids land application sites.

In the Chesapeake Bay watershed, nutrient reductions from sewage treatment plant upgrades were determined to be as cost-effective as, and more predictable than, the estimated reductions achieved through implementation of agricultural nonpoint source BMPs. The Chesapeake Bay Commission (2004) found average point source costs to remove N and P to be within the range of most widely implemented agricultural BMPs (USEPA, 2003b). The Commission stated that "this technology-based approach provides the highest degree of confidence for consistent, long-term reductions. Furthermore, the cost of this technology has continued to decline in recent years."

However, there are many differences in point source distribution, population, and income in various subbasins of the MARB compared to other areas of the country where point sources have had total N and P reductions (such as the Chesapeake Bay or Long Island Sound). Therefore, a cost-effectiveness analysis of point source controls of N and P in the MARB is needed to fully evaluate this particular method of reducing nutrient inputs to rivers in the context of nonpoint source control costs. A part of that analysis should consider the cost of N and P removal that could be optimized by establishing loading caps for individual treatment plants and/or groups of plants within river basins and by allowing nutrient credit trades between the plants. This "point-to-point" trading allows those plants that can most efficiently achieve reductions to sell nutrient reduction credits to plants that would incur much higher costs to achieve their loading cap. This approach is being used in Long Island Sound and in the Chesapeake Bay watershed within Virginia. These point-to-point trading programs are consistent with an overall cap and trade program as discussed in Section 4.4.3.

Another potential approach for reducing the nutrient discharge from sewage treatment plants, which could be applied alone or in combination with plant upgrades, is to encourage local sewer districts to establish more stringent nutrient pretreatment standards for private industries and other nondomestic sewer users. Meat packing, chemical manufacturing, and food processing are examples of the types of industries that generate wastewater containing large amounts of N and P. Through the regulatory authority granted to them under the National Pollutant Discharge Elimination System (NPDES) program, sewer districts can encourage industries to reduce their nutrient discharge to sewage treatment plants through the establishment of local sewer discharge nutrient limits as well as by the judicious development of technology-based wastewater surcharge rates.

The overall decrease in the mass of nutrients discharged into the local sewer system due to pretreatment will improve the quality of both the sewage treatment plant effluent and biosolids and will result in a net reduction of nutrients entering the MARB. A feasibility study is needed to evaluate the regulatory and economic options that could be applied to provide incentives for major industries to identify and implement pollution prevention measures to reduce and/or recycle nutrients that would otherwise be discharged into the local sewer system.

In addition, industrial treatment plant upgrades designed to remove nutrients can also reduce nutrients that are directly discharged to the MARB and the Gulf. Industrial discharges account for about 28% of the point source N flux and 23% of the point source P fluxes, or 75,000 metric tonne N/year (83,000 ton N/year) and 17,000 metric tonne P/year (18,700 ton P/year). Experience in other regions has shown that industrial sources could be targeted on a permit-by-permit basis since frequently a limited number of permitted facilities are responsible for a large part of the load. This approach could be recommended for the MARB. It would be useful to design initial efforts to focus on discharge categories likely to have high nutrient discharges. Examination of discharge information (Table 3.2, MART, 2006b) reveals that two categories (industrial organic chemicals and plastic materials/synthetic resins) account for about half of industrial N discharges, about 45,000 metric tonne N/year (50,000 ton N/year). For P, four categories (crude petroleum and natural gas, electrical services, refuse systems, and wet corn milling) account for about 40% of the industrial load or about 5,500 metric tonne P/year (6,000 ton P/year). Industries in these categories should be evaluated for opportunities to reduce N and P discharges through pollution prevention, process modification, or treatment.

While P removal is technologically feasible and widely implemented elsewhere, advanced treatment increases the amount of biosolids generated and, therefore, the land area needed to manage a given amount of biosolids based on P and N needs of the crop, rather than just the N requirements. This will create additional costs for biosolids-management programs in the MARB and needs to be considered when evaluating the total cost of implementing P removal at sewage and industrial treatment plants in the basin.

Unlike nitrogen, which can be biochemically transformed and removed from the sewage treatment plant as a volatile gas (N_2 and/or N_2O) through the nitrification/denitrification process, phosphorus is simply moved from the liquid to solid phases and accumulates in the biosolids. Physical upgrades in sewage treatment plants specifically aimed at reducing the phosphorus concentration in the effluent discharge typically include substantial additions of precipitating chemicals (e.g., alum) alone, or in combination with, higher efficiency membrane filtration. The net effect of these capital improvements is a significant increase in the mass of biosolids requiring handling and management. Most biosolids are beneficially used in crop production on land located as near to the treatment facility as feasible to minimize transportation costs. Transportation distances range from essentially zero to several 100 km depending on plant location, size, and the amount of biosolids or biosolid nutrient content. Phosphorus removal will increase both the mass of biosolids and the P content of the biosolids.

Biosolids application to agricultural land is regulated through the NPDES permit of the treatment facility. In many places in the MARB, land application of biosolids is based on the N needs of the crop. As with animal manures, biosolids application to meet crop N needs results in overapplication of P and buildup of bioavailable P in the soil surface. Research during the past two decades has indicated that soil P levels substantially in excess of crop needs can cause elevated P concentrations in runoff, particularly from critical source areas within fields. As a result, recommendations for application of organic nutrient sources, such as manure or biosolids, suggest that applications be limited based on P where the risk of loss is moderate to high. This will minimize the opportunity for P removed from discharged effluent to be lost in runoff when biosolids are land applied. All states now have a tool to estimate the potential for P loss from application of manure or biosolids. Nearly all states use a locally adapted version of the Phosphorus Site Index (PSI) to estimate P loss risk. Since biosolids currently contain more P relative to N than crops require, land application of biosolids should routinely involve an evaluation of the risk of P loss using the PSI or another risk assessment tool.

Key Findings and Recommendations

Sewage treatment plants and industrial dischargers represent a more significant source of N and P in the MARB than was originally identified in the *Integrated Assessment*. Tightening effluent limits on large sewage treatment plants together with establishing more stringent pretreatment nutrient standards on nondomestic sewer users may offer some of the most certain short-term and cost-effective opportunities for substantial nutrient reductions, particularly for P, but a full analysis of costs needs to be conducted in the context of nonpoint source reduction costs. Based on these findings, the Study Group offers the following recommendations.

- Tighter limits on N and P effluent discharge concentrations for major sewage treatment plants, together with concomitant reductions in nutrient discharges from nondomestic sewer users, should be considered, following an analysis of the cost and technical feasibility for a particular basin.
- A review of discharge data, including N and P loads, for industrial dischargers could identify possible industrial facilities to target for cost-effective reductions.
- Regulatory authorities should encourage or require sewage treatment plants to utilize phosphorus-based biosolids land application rates rather than the nitrogen-based rates in beneficial-use programs.

4.5.9 Ethanol and Water Quality in the MARB

The production of renewable fuels has been of interest since the 1973 oil price shocks, and technologies for the conversion of crops into ethanol and bio-diesel have existed since the 1940s. Currently about 99% of renewable transportation fuel produced domestically is ethanol from grains and oil crops, primarily corn (Institute for Agricultural and Trade Policy [IATP] 2006). This section focuses on the potential water quality implications of both ethanol production from corn and its potential production from lignocellulosic feedstocks.

The rapid growth in corn prices is primarily a result of increased energy prices (Kline et al., 2009). Increased ethanol production is only a minor contributor even though it is projected to rise from less to 2 billion gallons in 2001 to more than 19 billions gallons in 2009, a 950% increase (IATP, 2006). Current estimates are that about 75% of that production will be in the nine Upper Mississippi River Corn Belt states (IATP, 2006). The Food and Agricultural Policy Institute (FAPRI) projects that ethanol production from corn will increase from about 6.8 billion gallons in 2007 to over 14 billion gallons by 2012. Associated with this increase in ethanol production, FAPRI projects an increase in corn acreage from about 80 million acres to about 94 million acres in the same time period (www.fapri.missouri.edu). This growth of grain-based ethanol production may have major water quality implications for the MARB and the country.

Cellulosic ethanol is an alternative fuel made from a variety of nonfood feedstocks (such as agricultural residuals like corn stover and cereal straws, industrial plant byproducts like saw dust and paper pulp, and crops grown specifically for fuel production like switchgrass, *Panicum virgatum)*. By using a variety of regional feedstocks for refining cellulosic ethanol, the fuel can be produced in nearly every region of the country. Though it requires a more complex refining process, cellulosic ethanol produces less impacts on water quality, contains more net energy, and results in lower greenhouse emissions than traditional corn-based ethanol (Mclaughlin and Walsh, 1998). One of the challenges for wider use of cellulosic ethanol is that the cost of production is higher than current prices for corn ethanol and gasoline. Another challenge is that technology has not yet developed the fermentation efficiency for conversion of cellulosic feedstocks to the level at which it is commercially viable. Contributing to the high cost is the need to consolidate enough feedstock close to the plant to produce an adequate supply as well as the cost of transporting the heavy and bulky feedstock (Perlack and Turhollow, 2003).

Many hope that the heightened interest in biofuels will lead to a more sustainable mode of energy production by reducing impacts on water quality, recycling biomass residuals and emitting little, if any, greenhouse gases. The vision is that future biorefineries will use tailored perennial plants in increasing amounts (Perlack et al., 2005). Integration of agroenergy plant resources and biorefinery technologies can lead to a new manufacturing paradigm (Ragauskas et al., 2006). While these possibilities exist, much is unknown concerning how this future might develop and whether it is economically and technically viable.

4.5.9.1 Water Quality Implications of Projected Grain-Based Ethanol Production Levels

The Study Group could find no published estimates of the likely impact of the consequences of expanded corn-based ethanol production on nutrient flows from the MARB. To characterize the short-term potential impact, a set of simple calculations is reported in Table 4.8 that combine acreage projections from the FAPRI baseline for CRP and three major field crops in the Unites States with estimates of the per acre nutrient losses from these crops (CEAP, 2007). The second and third columns in the table report the projected nationwide acreage for the years 2007 and 2013 for corn, soybeans, wheat, and CRP and the fourth column reports the projected change in acreage for each. As can be seen, the FAPRI baseline projects a sizable increase in corn acreage, with that increase coming largely from soybeans and the CRP (totals do not add up since other cropland is omitted).

The fifth column estimates per acre N loss for corn, soybeans, and winter wheat based on the sum of waterborne losses reported in the CEAP assessment (http://www.nrcs.usda.gov/technical/nri/ceap/croplandreport/table 36, page 117) for the Upper Midwest region. The CEAP report did not estimate N loss from CRP, but for the current analysis, losses from CRP are assumed to be 10% of the average loss

Table 4.8 Estimated changes in N losses from cropping changes predicted by FAPRI from 2007 to 2013

	2007 FAPRI baseline (million acres)	2013 Acreage projections, FAPRI[a] (million acres)	Projected change in acreage[b] (million acres)	N Loss estimate per acre[c] (lbs./acre)	Difference in total N losses – million lbs[d]
Corn	78.3	93.7	15.4	28.1	431.6
Soybeans	75.5	67.9	−7.6	17.7	−134.2
Wheat	57.3	58.3	0.9	12.9	11.7
CRP	36.0	30.0	−6.0	2	−12
Total	247.2	249.9			297

[a]These projections are from the August, 2007 baseline http://www.fapri.missouri.edu/outreach/publications/2007/FAPRI_MU_Report_28_07.pdf
[b]This column is the difference between columns 1 and 2.
[c]Per acre estimates of N loss for corn, soybeans, and winter wheat are the sum of waterborne losses reported in the CEAP assessment (http://www.nrcs.usda.gov/technical/nri/ceap/croplandreport/ table 36, page 117) for the Upper Midwest region. The CEAP report did not estimate N loss from CRP, but for the current analysis, losses from CRP are assumed to be 10% of the average loss from cropland. The CEAP N loss rates are based on simulations using the Erosion Productivity Impact Calculator (EPIC) model. The CEAP estimates tend to overestimate surface losses and underestimate subsurface losses because EPIC does not estimate tile drainage losses that increase the dissolved subsurface loss of nitrate.
[d]The difference in total N losses is computed by multiplying the projected changes in acreage (column 3) by the N loss estimate per acre (column 4).

from cropland. The sixth column reports the estimated change in total N losses due to the change in acreage of CRP and each respective crop, with the sum in the bottom row representing the total projected increase in N loss. By this calculation, N losses nationwide could increase by 297 million pounds N/year between 2007 and 2013. Implications for nutrient loads to the Gulf of course depend on how much of the predicted acreage change will occur in the MARB. Assuming the MARB accounts for 80% of the change in cropping systems, additional losses of 238 million pounds N/year could be expected for the MARB.

While these estimates are rough and omit numerous factors that could affect the nutrient loss from these lands (policy changes like higher mandates for the ethanol content of gasoline, farming practices, energy prices, and climate change), they provide an idea of the magnitude of the possible short-term nutrient consequences from increased corn-based ethanol production.

4.5.9.2 Impacts on Nutrient Application to Corn

In the simple calculations made in Table 4.8, it was implicitly assumed that N application rates will remain unchanged. However, reductions in N application rates have been identified as one tool to reduce N loss from corn (CERN, 2000). The level of nitrogen application that maximizes farm profits for a given soil and climate is a function of price and input costs. Corn price has increased, but fertilizer N costs have also skyrocketed in recent years so it is not possible, without further analysis, to determine the net effects of these two price trajectories on fertilizer application rates. Further, as Laboski et al., (2008) point out, simply applying N at economically optimal rates will not resolve the issue of nitrate movement from fields in subsurface drainage, for nitrate losses occur in corn production systems even when no N is applied.

High corn prices associated with market impacts of increased ethanol production will make it less profitable for farmers to manage N conservatively. Higher corn prices are likely to reinforce the perception that assurance of adequate N is worth the cost, since farmers are more likely to be adverse to risks of yield loss when corn prices are high. Based on economic optimum yield and historic response to high corn prices by farmers, $4/bushel corn may tend to increase N application rates to levels where N use efficiency is lower. High corn prices also provide a disincentive for cropland retirement or conversion to perennials.

Finally, it is worth noting that a large literature exists on the likely magnitude of yield drag associated with continuous corn and other crop rotations. These effects may also mean higher fertilization over the levels assumed in the CEAP study used in Table 4.8. See Katsvairo and Cox (2000a, 2000b) and Pikul et al. (2005).

4.5.9.3 Grain Versus Cellulosic Ethanol and Water Quality

Cellulosic ethanol produced from perennial grasses, fast-growing woody species, manures, and other biomass residuals such as corn stover could allow the United States to meet renewable transportation fuel goals while improving water quality (Mann and Tolbert, 2000; Perlack et al., 2005). Yet the rapid expansion of grain-based ethanol products may be a disincentive to development of perennial crops or crop residual-based ethanol. The technology to produce ethanol from cellulosic materials is rapidly improving but is not yet operational. The production, storage, and handling infrastructure are in place for grain but not for perennial crops or residuals. Cellulosic material is harder to handle, and only biomass sources such as forestry residuals and corn stover are in sufficient abundance to provide reliable supplies.

Grain-based ethanol producers are interested in the development of technology using corn stover and other crop residue as feedstock. Crop residues represent the largest potential source of feedstock, projected to be 354 million metric tonne/year (390 million ton/year). Graham et al. (2007) estimated about 58 million dry metric tonne/year (64 million dry ton/year) could be removed with soil loss at “tolerable levels” (T) levels, but at ½ T soil loss removals could only be about 18 million metric tonne/year (19.8 million ton/year) (at 1995–2000 corn production levels). However, soil losses could increase 2–20 fold and still be below T. Therefore harvesting corn stover to keep soil losses just below T would result in substantial increases in erosion and associated N and P losses compared to current conservation or no-till production.

English et al. (2006) proposed that corn stover may be the largest potential source of cellulosic materials for ethanol production once cellulosic technologies are cost competitive. However, the contribution of returning stover to soil quality and quantity has long been recognized. Wilhelm et al. (2004) conclude that corn stover can be harvested for ethanol production, but recommendations for removal vary depending on regional yield, climatic conditions, and cultural practices.

Perennial grasses, including switchgrass and high biomass-producing trees, are currently considered the most promising energy crops (Kurt et al., 1998; McLaughlin and Kszos, 2005; Tolbert, 1998). Miscanthus and sweet sorghum have also been suggested as possible perennial feedstocks. This discussion focuses on switchgrass, which is a warm-season perennial native prairie grass that produces high biomass in its above ground growth and in deep roots. Switchgrass requires some N and P for optimal production, but less than corn. Switchgrass normally requires two growing seasons to become fully productive, but then it can grow for 20 years or more without replanting. Thus, either expected profitability from switchgrass production must be large enough to overcome early lower yields, or an incentive program will be needed to compensate the farmer during the 2-year transition. As mentioned previously, the transport and storage infrastructure needed to handle the large quantities of materials for an ethanol facility will need to be developed.

The evidence thus far suggests that switchgrass is a more favorable energy crop for reducing impacts on the land and climate; however, the technology for converting switchgrass to ethanol is not yet commercially viable. The fermentation co-product is a lignocellulosic material that can be dried and burned to provide part of the energy for the facility with net positive energy returns (Farrell et al., 2006). Switchgrass requires few nutrient additions, is not suited as a feed amendment, and can enhance threat to water quality. If it is grown instead of corn on productive soils, N and P losses are expected to be reduced by over 50% (Chesapeake Bay Program, 2003). Switchgrass will also sequester carbon, increase soil organic matter, and improve soil quality through its extensive, deep root system. These positive environmental attributes have substantial potential to provide multiple revenue streams. Lower production cost, greater net energy production, multiple revenue streams, and environmental benefits of switchgrass all favor its long-term use as a dedicated energy crop. However, the lag in development of fermentation technology and the lack of existing infrastructure prevent it from replacing corn as the major ethanol feedstock for the near future.

Increasing grain prices have increased the relative economic advantage that row crops, particularly corn, have over switchgrass. Substantial incentives will be needed before farmers would convert row crop land to switchgrass or other perennials at current market conditions. Babcock et al. (2007) estimated that the magnitude of subsidies would be significant and that conversion of all cropland to switchgrass in a watershed in northeastern Iowa would result in an 84, 83, 44, and 53% reduction, respectively, in sediment, total phosphorus (TP), nitrate (NO3), and total nitrogen (TN) at the watershed outlet compared to existing conditions. Model results also indicated that conversion of all cropland in the watershed to continuous corn would increase sediment, TP, NO3, and TN from current levels by 23, 128, 147, and 150%, respectively. They also evaluated the impact of growing switchgrass on all Highly Erodible Land (HEL) and continuous corn on other cropland. Careful placement of the switchgrass on other sensitive landscapes and as a buffer on non-HEL land could provide additional water quality benefits.

Key Findings and Recommendations

Expansion and intensification of corn production to support grain-based ethanol production and impacts of ethanol co-products from the animal production sector are likely to cause major increases in N and P losses in the MARB. The opportunity still exists to make choices that result in a renewable energy strategy that achieves energy goals with a reduced impact on the environment. Grain-based ethanol production is rapidly expanding, and the Study Group's preliminary calculations demonstrate a significant short-run increase in N and P losses to water resulting from current market incentives favoring corn.

Cellulosic ethanol production can be less environmentally detrimental, but current technology and infrastructures do not make it competitive with grain-based ethanol. Harvesting corn stover as a feedstock for cellulosic ethanol has water and soil quality implications. Switchgrass or other perennial grasses or woody biomass provide greater net energy and lower production costs and potentially higher total revenue with substantial environmental benefits when compared to corn and could become the dominant feedstock if investment, policy, and market conditions do not keep renewable energy policy focused on grain feedstocks. Based on these findings, the Study Group offers the following recommendations regarding biofuel production.

- Life-cycle analysis, examining all impacts to air, water, and climate, is needed to compare the various feedstocks for ethanol production.
- Research and development should focus on biofuel production systems that are both economically viable and ecologically desirable.
- If research continues to support the potential of cellulosic materials to meet energy and environmental goals, incentives (or the removal of disincentives) should be provided to promote ethanol production with more environmentally benign feedstocks.

4.5.10 Integrating Conservation Options

The previous sections have described land-management and conservation practices that can enhance nutrient loss reduction and water quality locally and in the

Gulf. As discussed, these practices vary, sometimes substantially, in their effectiveness among watersheds and subbasins in the MARB. Furthermore, there can be synergistic effects on nutrient loss reductions, where combinations of these practices can produce more (or less) than the sum of their individual reductions. In evaluating suites of management options, it is crucial to determine whether the nutrients that are not released to waters are being lost instead to other systems so that reactive N and P are not actually removed from the environment but just redistributed. These facts are an important part of the basis for our recommendation that watershed-based modeling approaches continue to be developed and that they be explicitly used to design optimal land-management systems within an adaptive management context. As noted in Sections 2.1.9 and 3.4, watershed-based models can be a key source of information for considering alternative sets of conservation practices and implementation approaches. Ideally, integrated modeling systems would be used to evaluate whether it is more cost-effective to reduce nutrient loadings with targeted nutrient management practices on the farm, to subsidize edge-of-field buffers in targeted watersheds, to change cropping patterns or to focus financing on well-placed off-site freshwater wetlands, or to implement some carefully chosen combination of these practices. However, while such models exist and are continuously being further improved, there remain limitations of these models in their current state (see Sections 2.1.9 and 3.4).

In Tables 4.9, 4.10, and 4.11, we provide a summary of the potential total nitrogen (TN) and phosphorus (TP) reduction efficiencies (percent, %) in surface runoff, subsurface flow, and tile drainage that can be realized where the various conservation practices could be implemented within the MARB. The cost-effectiveness of these measures will vary from site to site and with current and future land- and water-use designations. To a large extent, these estimates are based on relevant sections of this book and on reports by Devlin et al. (2003), Dinnes (2004), and Gitau et al. (2005). Where numeric values for reduction efficiency were not included in these reports, relative effects of practices were estimated based on expert opinion as negative (− , indicating increased export expected), positive (+, indicating reduced export expected), or neutral (±, indicating no significant effect expected).

Values for percent nutrient loss reductions are basin-scale averages, derived from edge-of-field and small watershed studies and not from widespread implementation. It must be emphasized that there is a great deal of site-specificity (spatial and temporal), which results in a wide range in observed conservation practice efficiency. While some of the conservation practices detailed have large, local, water quality benefits, they may not have a major impact on nutrient loss to the Gulf. To help facilitate implementation of practices that reduce nutrient loads to the Gulf, local water quality benefits are essential to MARB-wide adoption of these strategies. Estimates of N and P reductions are only appropriate to areas where a specific conservation practice can be implemented. For instance, it would not be effective to implement surface runoff control practices, such as sedimentation basins on flat lands with no concentrated surface flow of water. To a certain extent, N and P risk assessment tools that identify and quantify site vulnerability to N or P loss should be used at a local or field level to effectively target practices and to maximize reduction.

Table 4.9 Potential total nitrogen (TN) and phosphorus (TP) efficiencies (percent change) produced by *nutrient-use* conservation practices on surface runoff, subsurface flow, and tile drainage. Estimates are average values for a multiple-year basis, and some of the numbers in this table are based on a very small amount of field information. Shading highlights the methods producing the greatest reduction efficiencies within the three types of N and P loss (surface runoff, subsurface, and tile drainage)

	Surface runoff		Subsurface flow		Tile drainage	
Conservation practice	TN	TP	TN	TP	TN	TP
Nitrification and urease inhibitors	+[a]	±	+	±	1–21[b]	±
Nitrogen: spring versus fall application	+	±[d]	0–25[c]	±	10–30 %[b]	±
Nitrogen: Recommended rate versus above-recommended rate	28–44[b]	+[d]	+	±	27–50[b]	±
Nitrogen: Subsurface versus surface broadcast	50[e]	20[f]	–	±	16[b]	±
Phosphorus: Avoid runoff producing rainfall	±	28–57[b]	±	+	±	+
Phosphorus: Rate balanced to crop use versus above-recommended rate	0–25[e]	15–47[b]	±	36[b]	±	25[b]
Phosphorus: Subsurface versus surface broadcast	±	8–92[a,b]	±	–	±	–
Manure: bioenergy, treatment, alternative use, transport to nutrient-deficit areas	+[f]	+[f]	+[f]	+[f]	+[f]	+[f]
Adoption of comprehensive farm nutrient management plan	0–65[e,g]	0–45[e,g]	+[f]	+[f]	+[f]	+[f]

NOTE: For references, see Table 4.11.

Implementation of any one of the tabulated conservation practices can positively or negatively influence the effectiveness of another.

Awareness of the weather forecast in planning any nutrient application or tillage operation is important to avoiding rainfall-induced runoff of applied nutrients and erosion. The conversion of cropped acres to perennial crops is distinguished from conversion to CRP lands, in that perennial crops will include grasses harvested for cellulosic biofuel production, which may receive maintenance or low fertilizer N and P inputs. The conversion of lands to CRP and from annual cropping to perennials is expected to decrease N and P loss because of reduced fertilizer and manure nutrient inputs and reduced erosion afforded by increased vegetative cover. Improved N use efficiency via appropriate timing, rate, and method of application is expected to benefit P loss reductions by increasing crop P uptake and removal (if harvested).

Table 4.10 Potential total nitrogen (TN) and phosphorus (TP) efficiencies (percent change) produced by *in-field* conservation practices on surface runoff, subsurface flow, and tile drainage. Estimates are average values for a multiple-year basis, and some of the numbers in this table are based on a very small amount of field information. Shading highlights the methods producing the greatest reduction efficiencies within the three types of N and P loss (surface runoff, subsurface, and tile drainage)

	Surface runoff		Subsurface flow		Tile drainage	
Conservation practice	TN	TP	TN	TP	TN	TP
No-till versus conventional tillage	0–25[b,e]	35–70[b,e]	–	±	–	±
Cover crops	50[b]	7–63[b]	+	48[b]	13–50[b]	+
Diverse cropping systems and rotations within row cropping [g,h]	25–70[b,e,g]	25–88[b]	±	±	52–93[b]	±
Contour plowing and terracing	20–55[c,f]	30–75[e,g]	–	±	±	±
Standard tile drainage versus undrained	25[g]	70[g]	+	±	–	–
Water table management versus uncontrolled drainage	–	–	+	+	25–54[b]	+
Shallow and/or wide versus standard tile placement	–	–	+	+	39[b]	25–42[b]
Conversion to CRP	40[b]	+	40[b]	+	40–97[b]	+
Conversion to perennials crops	+60–90[h]	+75–95[h]	+90[i]	+	+	+
Livestock exclusion from streams versus constant intensive grazing	10–80[b,g]	32–76[g,i,k]	+	75[b]	±	±
Managed grazing versus constant intensive grazing	–100–80[b,g]	0–78 [b,g]	+	+	+	±
In-field vegetative buffers	12–51[b,e,g]	4–67[b,e,g]	±	±	–	–

NOTE: For references, see Table 4.11.

The estimated reduction efficiencies in Tables 4.9, 4.10, and 4.11 are based on edge-of-field losses for studies conducted within the MARB and do not represent expected whole-basin reductions. These values represent potential reductions only for those areas where the particular practices could be implemented and do not address how broadly a practice could be applied. The shaded areas indicate those practices expected to have the greatest impact on reducing nutrient export from the MARB as a whole: red shading indicates conservation practices that translate into N loss reduction in tile drainage, green shading is for surface runoff of N and P, and blue shading for nutrient loss in subsurface flow. It is clear that where edge-of-field loss estimates are available, there is a large variability in reduction efficiencies, which is both temporally and spatially dependent. This inherent variability must be recognized when developing conservation or remedial strategies for the MARB,

Table 4.11 Potential total nitrogen (TN) and phosphorus (TP) efficiencies (percent change) produced by *off-site* conservation practices on surface runoff, subsurface flow, and tile drainage. Estimates are average values for a multiple-year basis, and some of the numbers in this table are based on a very small amount of field information. Shading highlights the methods producing the greatest reduction efficiencies within the three types of N and P loss (surface runoff, subsurface, and tile drainage)

	Surface runoff		Subsurface flow		Tile drainage	
Conservation practice	TN	TP	TN	TP	TN	TP
Sedimentation basins	55[g]	65[g]	±	±	±	±
Riparian buffers: total n total P	50–82[g,i]	40–93[g,i,k]	+	+	±	±
Riparian buffers: nitrate-N and dissolved P	41–92[i,k]	28–85[i,k]				
Wetlands: total P	61–92[b,g]	0–79[g,i,k]	9–74[b]	+	20–90[i,k]	+
Wetlands: dissolved P		22–86[g,i,k]				

[a]Relative effects of practices estimated based on expert opinion as negative (– , indicating increased export expected), positive (+, indicating reduced export expected), or neutral (±, indicating no significant effect expected).
[b]From Dinnes (2004) report or from Study Group report. Values from IA, IL, MO, MN, NE, OH, and OK are included.
[c]From Randall and Sawyer (2005), nitrogen application timing, forms, and methods. pp. 73–84. Session 6, UMRSHNC (2006) report.
[d]Increased crop yields afforded with N fertilizer is likely to increase P uptake by crop and P removal if harvested.
[e]From Devlin et al. (2003).
[f]Improved manure management leads to lower land application and thereby less potential for loss in any pathway.
[g]Values based on data included in Gitau et al. (2005).
[h]Studies with only corn–soybean systems are not included, although they were included in Dinnes (2004).
[i]Values from Smith et al. (1992).
[j]Values from Randall et al. (1997).
[k]Values are modifications of values in Dinnes (2004) based on values in Study Group report.

in the context of probability of expected outcomes. It is also a key component of the conservation premise that there is no "one size fits all" rationale for adaptive management.

As a complement to the information summarized in Tables 4.9, 4.10, and 4.11 a second summary of the likely environmental benefits is provided in association with the conservation and land management. In Tables 4.12 and 4.13, the focus is on the broader contribution these practices can have with respect to a wide variety of environmental services including local water quality, carbon sequestration in agricultural soils, wildlife habitat, biodiversity, general recreational activities, and air pollution. These effects are based on the scientific literature and professional judgment, and potential repercussions are indicated only as being positive (+) or negative (–) or having no effect (0).

Table 4.12 Anticipated benefits associated with different agricultural management options

Agricultural management option	Reduce N load to Gulf	Reduce P load to Gulf	Local surface water quality		Ground-water quality	Carbon sequestra-tion	Local wildlife habitat[a]	Biodiversity[a]	Recreational activities	Air pollution reduction	Soil quality
			N	P and sediments							
Decrease drainage intensity	+	–	+	–	0	0	+	+	+	0	0
Increase freshwater wetlands	+	+/?	+	+/?	0	+	+	+	+	–	0
Forested riparian buffers	+	+	+	+	+	+	+	+	+	+	+
Herbaceous riparian buffers	+	+	+	+	+	+	+	+	+	+	+
Improve manure mgmt.	+	+	+	+	+	0	0	0	0	+	+
Increase acreage of perennials	+	+	+	+	+	+	+	+	+	+	+
Increase acres of farmland retired	+	+	+	+	+	+	+	+	+	+	+
Reduce fertilizer N and/or P application	+	+	+	+	+	0	+	+	+	+	0
Spring fertilizer N and/or P application	+	0	+	0	+	0	0	0	0	0	0
Expand corn-based ethanol production	–	–	–	–	–	–	–	–	–	–	–
Expand cellulosic ethanol production	+	+	+	+	+	+	+	+	+	+	+

+ = will lead to improvements in conditions; – = likely to be further degraded; 0 = will have little effect; ? = effect unknown.

Table 4.13 Anticipated benefits associated with other management options

Management option	Reduce N load to Gulf	Reduce P load to Gulf	Local surface water quality		Groundwater quality	Carbon sequestration	Local wildlife habitat	Biodiversity	Recreational activities	Air pollution reduction	Soil quality
			N	P and Sediments							
Decrease NO_x emissions	+	0	+	0	0	0	0	0	0	+	0
Reduce point source loads	+	+	+	+	0	0	+	+	+	0	0
Reduce urban nonpoint source loads	+	+	+	+	+	0	+	+	+	0	+
Enhance floodplain connectivity	+	+	+	+	0	+	+	+	+	0	0
Atchafalaya diversion	?	?	?	?	0	0	0	0	?	0	0
Increase coastal wetlands	?	+	?	?	0	+	+	+	+	0	+

+ = will lead to improvements in conditions; – = likely to be further degraded; 0 = will have little effect; ? = effect unknown.

In each of these tables, the effects predicted assume that conservation practices are implemented and managed (maintained) as designed to maximize effectiveness and life expectancies. Inadequate implementation and maintenance can lead to poor performance of such systems. Further, these strategies need to be carefully targeted at an appropriate level of intensity and over sufficient time in order to effectively reduce nutrient export.

Finally, when considering these tables, it is important to note there are synergistic effects of combinations of conservation practices that result in greater nutrient loss reductions than do individual practices (Table 4.10). For example, N application management that minimizes the potential for excess N available to be leached (nutrient management, Table 4.9) should be combined with efforts to reduce the potential off-site movement of water (infield management, Table 4.9). Conversely, there are potential trade-offs. For example, reduced-till, no-till, and tile drainage can decrease runoff, erosion, and P loss but can enhance NO_3 nitrate leaching potential. As another example, while N-based manure application can be a cost-effective N source to meet crop N needs, P may be overapplied, increasing the potential for increased runoff and loss of P.

Key Findings and Recommendations

A number of conclusions concerning the appropriate use of conservation practices can be drawn from these tables. First, there is no "one size fits all" land-use or conservation practice strategy that will be cost-effective in all locations. Rather, site-specific and regional optimization of conservation practices and appropriate targeting of conservation practices and measures will be needed and will include a broad range of alternative practices and land uses, such as crop, animal, fertilizer, and drainage management measures targeted to appropriate areas. The reduction efficiencies of these practices are spatially and temporally variable, making it impossible to assign a specific reduction efficiency for any given conservation practice. As information from ongoing monitoring of nutrient loss reduction efficiencies becomes available, we will be better be able to determine what major factors influence reduction efficiencies. This learning and integration of new knowledge is important and will enhance the process of adaptive management.

Second, practices that are likely to address NGOM hypoxia effectively in tile-drained landscapes can differ markedly from those appropriate in nontiled lands. Further, while there are no-one-size-fits-all strategies, there are some approaches that appear particularly promising. For example, inter-seeding of leguminous cover or relay crops within corn and other grain rotations can decrease fertilizer N requirements, reduce soil profile N at critical loss times of the year, and mine excess soil P. Reconnecting the floodplain with managed agricultural lands, by managing hydrology to increase the amount of time water is retained on the land (wetland) prior to entering the major fluvial systems, should be considered an important part of an adaptive management plan to reduce NGOM hypoxia.

Third, practices that are likely to be cost-effective in addressing NGOM hypoxia may not be the same that yield the highest benefits in other environmental dimensions. This has important planning and implementation implications, for it suggests that, when considering implementation strategies, the optimal set of conservation practices and sinks needs to be considered with respect both to NGOM hypoxia and to the suite of other environmental concerns that are likely to vary regionally.

Finally, in considering information from the tables and "optimal" sets of practices, the principles of adaptive management imply that approaches need to be changed and updated with time to maximize overall efficiency. In the process, more information can and will be learned about the effectiveness of these practices. This information can be used both to improve the performance of water quality models to aid in better implementation strategies and directly to improve targeting of conservation practices and actions. Based on these findings, the Study Group offers the following recommendations.

- There is great temporal and spatial variability in nutrient loss reduction efficiencies of the various conservation practices available. Thus, continued, new, and enhanced small watershed-based studies of suites of conservation practices as applied in the real-world are necessary and should be set in a context of research, monitoring, and demonstration to stakeholders so that progress (or lack thereof) in response to management change can be assessed. A variety of response measures relevant to different watershed scales and environmental concerns should be monitored. These measures should include both performance measures (e.g., nutrient loading at subwatershed levels, estimates of carbon sequestered on the landscape), and practice-based measures (e.g., number of acres of wetlands installed, miles of conservation buffers installed).
- To reduce spring nitrate loss from tile-drained regions, alternative and more complex cropping systems (including perennials) are thought to be the most effective method of reducing losses. However, given current constraints in cropping systems, the Study Group recommends reducing or discontinuing fall N application for corn, improved N fertilizer management techniques, use of cover crops, wetland establishment, and drainage management where appropriate.
- For P loss reduction, the Study Group again finds that alternative and complex cropping systems are most effective. For current cropping systems, the Study Group recommends that riparian buffer strips; improved P fertilizer and manure management; and, where appropriate, cover crops be implemented.

- Where appreciable drainage occurs in the fall and winter, controlled drainage could significantly reduce nitrate losses but can be expected to increase surface runoff and losses of particulate contaminants.
- If precision agriculture and controlled-release fertilizer technologies are proven to provide reductions in losses of N and P to water resources, then incentives should be considered to stimulate their adoption.
- Incentives for conversion to perennials, which have potential future use as cellulosic biofuels production, should be established to promote the co-benefit of greatly reduced nitrate and P loss from agricultural systems.
- There should be a focus on conservation practices and implementation strategies that appropriately match the nutrient reduction strategies with the goals of reducing NGOM hypoxia as well as local/regional environmental goals (carbon sequestration, wildlife, air quality, local water quality, etc.). Given the breadth and magnitude of these additional environmental goals, these "co-benefits" should be incorporated in the planning process.
- Information on effectiveness and geographic appropriateness of various conservation practices and nutrient reduction strategies should be used in conjunction with formal models to plan implementation strategies for conservation measures that effect a reduction in nutrient loading to the NGOM.