Science for Solutions

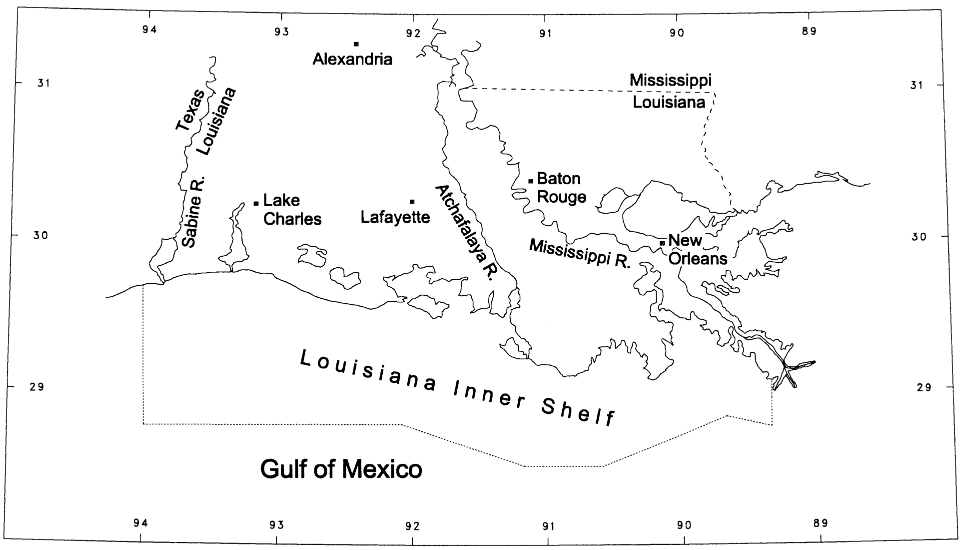
NOAA COASTAL OCEAN PROGRAM
Decision Analysis Series No. 18

Effects of Reducing Nutrient Loads to Surface Waters within the Mississippi River Basin and the Gulf of Mexico

Topic 4 Report for the Integrated Assessment on Hypoxia in the Gulf of Mexico

Patrick L. Brezonik, Victor J. Bierman, Jr., Richard Alexander, James Anderson, John Barko, Mark Dortch, Lorin Hatch, Gary L. Hitchcock, Dennis Keeney, David Mulla, Val Smith, Clive Walker, Terry Whitledge, and William J. Wiseman, Jr.

May 1999



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

GULF OF MEXICO HYPOXIA ASSESSMENT

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

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Cover image: Location map of the study area for the water quality model.

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May 1999

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List of Abbreviations and Acronyms

BOD	Biological oxygen demand
CBOD	Carbonaceous biochemical oxygen demand
CENR	Committee on Environment and Natural Resources
DO	Dissolved oxygen concentration
EPA	U.S. Environmental Protection Agency
IMPs	Improved management practices
LIS	Louisiana Inner Shelf portion of the northern Gulf of Mexico
LMR(B)	Lower Mississippi River (Basin)
LSU	Louisiana State University
LTI	Limno-Tech, Inc.
LUMCON	Louisiana Universities Marine Consortium
MAR	Mississippi–Atchafalaya River
MRB	Mississippi River Basin
N	Nitrogen
N:C	Nitrogen-to-carbon ratio
NBOD	Nitrogenous biochemical oxygen demand
NECOP	Nutrient Enhanced Coastal Ocean Productivity
NOAA	National Oceanic and Atmospheric Administration
N:P	Nitrogen-to-phosphorus ratio
P	Phosphorus
P:C	Phosphorus-to-carbon ratio
POC	Particulate organic carbon
PON	Particulate organic nitrogen
SOD	Sediment oxygen demand
TN	Total nitrogen concentration
TP	Total phosphorus concentration
UMR(B)	Upper Mississippi River (Basin)
USGS	U.S. Geological Survey
WASP4	Water Analysis Simulation Program, Version 4

Foreword

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

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

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

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

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

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

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

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

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

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

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

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

David Johnson, Director
Coastal Ocean Program

Donald Scavia, Chief Scientist
National Ocean Service

Executive Summary

The overall goal of this assessment was to evaluate the effects of nutrient-source reductions that may be implemented in the Mississippi River Basin (MRB) to reduce the problem of low oxygen conditions (hypoxia) in the nearshore Gulf of Mexico. Such source reductions would affect the quality of surface waters—streams, rivers, and reservoirs—in the drainage basin itself, as well as nearshore Gulf waters. The task group's work was divided into addressing the effects of nutrient-source reductions on: (1) surface waters in the MRB and (2) hypoxia in the Gulf of Mexico.

MISSISSIPPI RIVER BASIN

The freshwater phase had two objectives:

- evaluate the effects of nutrient-source reductions in the drainage basin on nutrient concentrations and loads in flowing waters of the basin; and
- describe the effects of changes in nutrient concentrations, and evaluate the magnitude of those effects, on ecological and related water quality conditions in these flowing waters.

A comprehensive approach was taken: both major nutrients (nitrogen and phosphorus) for plant growth and a wide range of water quality factors were evaluated. Because of the enormity of the basin and the complexity and diversity of its aquatic ecosystems, it was not possible to assess site-specific effects. The analysis focused on areas with the highest concentrations and loads of nutrients—the agriculturally dominated central region of the basin, and especially the Corn Belt.

The largest source of nutrients in the MRB is agricultural activity, but point sources, urban runoff, and atmospheric deposition also contribute. Options for reductions from each major category of nutrient sources were reviewed, with special attention to on-farm management practices for agricultural sources. Yield information (mass lost per unit area per time) for nitrate (a soluble form of nitrogen (N) that in high concentration is associated with degraded water quality) was used to assess the need for improvements in agricultural practices. Nitrate yields in the MRB generally are highest in the Corn Belt and lowest in highly forested watersheds and arid sub-basins. The processes of delivering nutrients to receiving waters differ for nitrogen and phosphorus (P). The former is especially associated with subsurface drainage, and artificial (tile) drainage is an important factor; the latter is associated with surface runoff, and soil erosion is an important factor. Consequently, options for reducing losses also must be considered separately.

A case study on the Minnesota River Basin illustrated the potential effectiveness of improved management practices (IMPs) on losses from fields. Because climatic, soil, and cropping conditions are generally similar throughout the Corn Belt, improvements described in the case study apply throughout the Corn Belt. Significant reductions in losses can be achieved by such IMPs as: increasing the spacing of tile drainage; controlling water table levels to promote denitrification in soil; routing tile drainage through wetlands; planting strips of grass and forest as buffers; changing from row- to perennial-cropping systems; planting a cover crop during the fall and winter; switching from conventional to reduced tillage; switching from fall to spring application of fertilizer; and limiting fertilizer and manure applications to agronomically recommended rates. Some IMPs reduced losses by only 10–20% of baseline conditions, but others reduced losses by up to 90% in field studies. From a practical standpoint, not all the options are equally viable.

Significant load reductions at the scale of watersheds will require widespread adoption of IMPs. It is important to recognize that experiences at selected sites and small watersheds should not be linearly extrapolated to estimate changes in nutrient deliveries and transport over large areas. In particular, experiences with cropland watersheds on relatively level land with highly developed tile drainage do not provide evidence for equivalent changes in nutrient loadings over large drainage basins with multiple land uses, variable slopes, and long river systems. Moreover, human responses to changes in agricultural practices tend to be buffered as well. For example, imposing restraints on fertilizer and manure applications in targeted areas will reduce some agricultural production in those areas, raise the prices of the affected commodities, and induce farmers elsewhere to increase production, with associated increases in nutrient use. These effects can be estimated only by multidisciplinary studies of the large areas.

The extent of reductions achieved depends on site-specific characteristics (climate, soils, cropping history), the types of improvements in management, and the baseline conditions to which the management improvements are being compared. Among the controllable measures, fertilizer application rate is a major factor, but the effectiveness of reducing rates depends on climatic conditions and the extent to which baseline applications exceed recommended rates. For example, a combination of modeling and field studies in southern Minnesota showed that incremental loss rates approached 100% of the nitrogen in added fertilizer when application rates were excessive (> 130 kg/ha) and annual rainfall was above normal. Losses were much smaller and nearly independent of rainfall when applications were at or below recommendations based on agronomic production.

Aside from the on-site management practices for diffuse nutrient sources that are described in this report, several other large-scale strategies exist for decreasing nutrient export from the MRB. These include changes in land use (e.g., conversion of cropland to conservation areas) and re-establishment of some portion of the large acreage of wetlands in the MRB that have been drained and converted mostly to agricultural lands over the past 150 years. These important approaches are described in the report of another task group (Mitsch et al. 1999) and are not discussed in detail in this report.

Not all nutrients entering MRB flowing waters are transported to the Gulf; nutrient retention and loss occur by denitrification, sedimentation, and plant uptake. Loss and retention decrease the downstream transport of nutrients and the impact of nutrient use in the upper drainage basin on downstream water quality. As a result, reductions in the downstream load of nutrients will be less than any reductions in the mass of fertilizer applied to fields. Analysis of water quality data shows that substantial processing of nutrients occurs in the rivers by primary production, but data are inadequate to determine the amounts of nutrients lost or retained in the rivers by this process. Model-based estimates were made for in-stream nutrient losses in small tributaries and large rivers of the Upper and Lower Mississippi River Basins using literature-based estimates of nutrient loss rates in rivers and mean water travel times in the rivers. The mean percentage loss of total nitrogen (TN) was estimated to be ~35–40% in small tributaries and ~20% in mainstem rivers. In contrast, the mean loss of total phosphorus (TP) was estimated to be ~28–37% in small tributaries and negligible in the mainstem channels. The latter results are reasonable for the free-flowing main channels, but it is likely that there is some TP retention (as yet unquantified) behind dams in the Upper Mississippi River.

To address the potential benefits of lower nutrient concentrations on ecosystem and water quality, we examined: (1) the potential for decreased frequency of violations in water quality standards related to nutrient conditions in MRB waters; (2) potential reductions in biochemical oxygen demand; (3) decreased frequencies of exceeding nutrient concentration criteria related to river eutrophication; (4) effects on plankton composition and production and on nuisance algal blooms; (5) effects on macrophyte communities; and (6) effects on fish communities.

Violations of water quality standards for dissolved oxygen, pH, nitrate, and un-ionized ammonia in general are uncommon under current conditions, but violations are more frequent at some sites. Review of state and EPA 305(b) assessments also indicates that most MRB states have substantial numbers of river miles that suffer use impairment related to nutrient conditions or do not fully support three resource uses (aquatic life support, fish consumption, and swimming). Excessive levels of nutrients are a major reason why many river miles in the MRB do not fully support these uses, especially the swimming and aquatic life-support uses, and reductions in nutrient concentrations, if significant, can improve water quality in these river stretches. Present levels of biological oxygen demand (BOD) are not high enough to cause low dissolved oxygen, except perhaps in isolated portions of MRB rivers and streams. Reduced nutrient concentrations would lead to somewhat lower BOD levels in the waters, but would not significantly change dissolved oxygen levels in the rivers.

About 30–55% of the hydrologic cataloging units (HCUs) of the Ohio, Lower Mississippi, and Tennessee sub-basins exceed a proposed eutrophic criterion for TP in flowing waters, and 16–40% of the HCUs in these regions exceed the proposed flowing-water criterion for TN. Higher exceedance frequencies were found in the Missouri, Upper Mississippi, and Arkansas–Red sub-basins (~80% of the HCUs for TP and 70–75% for TN). A regression-based model showed that a 30% reduction in median TP concentrations is required in the Upper Mississippi, Arkansas–Red, and Missouri regions to obtain a 10% reduction in the HCUs that exceed the trophic criterion for TP. In contrast, only a 15% reduction is required in the Ohio, Tennessee, and Lower Mississippi regions to achieve a 10% reduction in the rate of exceedance.

Recently published data for many rivers show that chlorophyll concentrations are correlated with TP concentrations. Substantial scatter exists in the relationship, but the general implications are clear: other factors being equal, phytoplankton biomass increases with increased P in rivers, just as it does in lakes. Analysis of nitrogen-to-phosphorus (N:P) ratios in waters across the MRB indicated that 69% of the waters fell into the combined N+P and P-limited class, and 31% of the sites exhibited potential N-limitation.

An empirical relationship was used to predict the improvements in chlorophyll that would occur *on average* following reductions in TP concentrations in river reaches of fixed catchment size. Many factors besides TP concentrations influence algal production. In addition, large responses of chlorophyll to reductions in TP should not be expected when inorganic P values are high. In such cases, substantial loading reductions may be necessary to induce a measurable response in algal biomass at a given site.

Aquatic macrophytes have important effects on water quality in shallow systems. If reductions in N and P levels increase underwater light, the distribution of aquatic macrophytes will expand in the Upper Mississippi River, with concomitant beneficial effects on water quality. Increased macrophyte abundance may increase nutrient retention within the river system (because of enhanced deposition and retention of suspended sediment), leading to lower delivery rates of nutrients to the Gulf of Mexico than would otherwise be predicted from direct effects of external source reductions alone. Increased macrophyte abundance also would increase the habitat for fish. Several potential negative effects of decreased nutrient loading were examined, including lower fish production, and none was found likely to cause significant impacts at the loading reduction levels likely to occur.

GULF OF MEXICO

The goal of the Gulf of Mexico portion of this assessment was to investigate whether water quality in the Louisiana Inner Shelf (LIS) portion of the northern Gulf of Mexico would be responsive to changes in nutrient loadings from the Mississippi–Atchafalaya River (MAR). The specific objectives were to investigate whether dissolved oxygen and chlorophyll concentrations are sensitive to changes in MAR nitrogen and phosphorus loadings, and to estimate the magnitudes of potential reductions in these loadings that may be necessary to improve present water quality conditions, especially seasonal hypoxia. The purpose of this assessment was not to establish target nutrient loading objectives, but rather to determine the range of reductions in nutrient loadings that may need to be evaluated in future studies.

The study's objectives were met by conducting forecast simulations with a quantitative water quality model. It was found that dissolved oxygen and chlorophyll concentrations on the LIS appear to respond to reductions in nutrient loadings from the MAR; however, there are significant uncertainties in the magnitudes of these responses for a given nutrient loading reduction. These uncertainties are due to lack of information on controlling physical, chemical, and biological processes, and to natural variability in hydrometeorological conditions in the northern Gulf of Mexico.

For reductions in nutrient loadings of 20–30%, bottom-water dissolved oxygen concentrations were estimated to increase by 15–50%, and surface chlorophyll concentrations were estimated to decrease by 5–10%. The ranges correspond to different assumptions for sediment responses and large-scale Gulf of Mexico water quality, and to different hydrometeorological conditions among different years. Although differences in responses between reductions in nitrogen and phosphorus loadings were generally significant, there was a tendency for responses to be somewhat greater for nitrogen reductions than for phosphorus reductions, especially for dissolved oxygen.

A significant obstacle to reducing uncertainties in quantifying linkages between MAR nutrient loadings and water quality responses in the northern Gulf of Mexico is lack of a sufficiently comprehensive database. There is a basic need for good physical oceanographic data on water movements and other physical processes. There is also a need for data on chemical and biological processes that influence hypoxia. Although the existing database is comprehensive in many respects, the data were acquired primarily to characterize water quality responses, not to provide data for quantifying load–response relationships or principal controlling processes. To accomplish this objective, a conceptual model of ecosystem structure and function should be created and used as a foundation on which to develop future monitoring plans.

Field data generated by a comprehensive monitoring program are necessary but not sufficient for developing and validating quantitative models. There is not yet a complete understanding of the physical, chemical, and biological processes that influence water quality responses in the northern Gulf of Mexico. Research is needed to better understand these processes and to provide information for representing them in quantitative models. The most important of these processes involves water circulation; stratification; primary productivity; underwater-light attenuation; the influence of phytoplankton dynamics on fate pathways for organic carbon; and cycling and transformation of nutrients, carbon, and oxygen.

The results presented in this report are preliminary results from an ongoing research program, and should be considered provisional in nature. To reduce uncertainties in these results, future modeling work should include linkage of the water quality model with a hydrodynamic model of Gulf of Mexico circulation, expansion of the model's spatial domain, and refinement of the model's horizontal and vertical spatial resolution. The water quality model itself should be expanded to include a sediment diagenesis submodel, multiple phytoplankton groups, and silica as a potential limiting nutrient.

CHAPTER 1

Introduction

1.1 SCOPE OF THIS REPORT

The general charge to Task Group 4 was to assess the effects of nutrient-source reduction techniques that may be implemented in the Mississippi River Basin to reduce the problem of low oxygen conditions (hypoxia) in the nearshore Gulf of Mexico. Although the source reductions would be directed at solving a problem that is external to the river basin itself, they would also affect the quality of surface waters—streams, rivers, and reservoirs—within the drainage basin. Therefore, the task group's work was divided into addressing the effects of nutrient-source reduction on: (1) the fresh waters in the MRB and (2) hypoxia in the Gulf of Mexico. Separate groups were assembled to address each part. The organization of this report reflects this division.

Chapter 2 of this report describes the Mississippi River Basin and the coastal region of the Gulf of Mexico where hypoxia occurs, the sources of data and approaches used in the two assessment phases, and the model and model assumptions used for the Gulf of Mexico phase. Chapter 3 presents the results of the assessments and discusses and interprets their implications. Chapter 4 describes further research needs identified by the task group and recommends additional monitoring and development of simulation models. Finally, Chapter 5 summarizes the major conclusions from both parts of the study.

The purpose of this analysis was not to establish target nutrient loading objectives, but to determine the range of nutrient loading reductions that may need to be evaluated in future studies. The results presented here are preliminary results from an ongoing research program and should be considered provisional in nature.

1.1.1 Mississippi River Basin

The goal of the freshwater part of this study was to evaluate the likely effects of nutrient-source reductions on nutrient concentrations and loads in flowing waters of the MRB and to assess the effects of these reductions on the overall quality and ecosystem integrity of these waters.

Because the six task groups addressing the hypoxia issue conducted their assessments simultaneously, some overlap with the responsibilities of other task groups was necessary in order for our task group to conduct its own assessment. In particular, Task Group 3 had primary responsibility for assessing nutrient sources to the Gulf, and in so doing addressed questions of nutrient retention and losses within the drainage basin. For our task group to evaluate the effects of changes in land-management practices on nutrient concentrations in the streams and rivers, we also needed to evaluate the extent of nutrient retention and losses within the MRB. Similarly, Task Group 5 assessed the various options for nutrient-source reduction in the MRB, but this question was of key importance in our own analysis of the magnitude of reductions we could expect in nutrient concentrations and loads in the rivers. Consequently, our task group also addressed this question, focusing especially on the potential and opportunities for reductions from agricultural sources.

Regarding the effects of reductions in nutrient concentrations on river water quality, the task group took as broad and quantitative an approach as possible, consistent with the availability of data on the MRB freshwater system and our present state of knowledge about river eutrophication. We considered effects of lower nutrient concentrations on other aspects of chemical water quality, including effects on violations of water quality standards for dissolved oxygen and other nutrient-related variables. We also considered effects on a wide range of potential biological conditions, including planktonic, macrophyte, and fish community composition and production.

1.1.2 Gulf of Mexico

The goal of the Gulf of Mexico portion of this study was to investigate whether water quality parameters in the Louisiana Inner Shelf (LIS) portion of the northern Gulf of Mexico are responsive to changes in nutrient loadings from the Mississippi–Atchafalaya River (MAR). Task Group 1 (Rabalais et al. 1999) presented a comprehensive description of hypoxia in the northern Gulf of Mexico and its relationship to nutrient loadings. Task Group 2 (Diaz and Solow 1999) investigated the broader ecological consequences of hypoxia, including its potential impacts on benthos and the fisheries. The principal water quality variables of interest in this report's task group assessment were dissolved oxygen concentrations in bottom waters and chlorophyll concentration in surface waters. The principal nutrients were nitrogen and phosphorus.

This analysis focused on the question of what level of MAR nutrient load reductions may cause a change in LIS water quality. Specifically, it was of interest to investigate whether reductions of 20–30% were sufficient to produce a water quality response, or whether reductions of up to 70% may be required. An answer to this question is crucial in determining whether reducing MAR nutrient loadings is a feasible option for improving present water quality conditions, especially seasonal hypoxia. With respect to achievability, Task Group 5 (Mitsch et al. 1999) concluded that greater than 50% of the nitrogen loading to the Gulf of Mexico could be reduced by implementing a number of proven techniques working in concert.

Within this context, the present analysis had the following specific objectives:

- investigate whether dissolved oxygen and chlorophyll concentrations are sensitive to changes in MAR nitrogen and phosphorus loadings, and
- estimate the magnitudes of potential reductions in nitrogen and/or phosphorus loadings that may be necessary to improve present water quality conditions, especially seasonal hypoxia.

CHAPTER 2

Site Description and Methods

This chapter consists of two major sections. The first describes the landscape characteristics of the Mississippi River Basin (MRB) and briefly discusses the approach and data sources used to assess the likely effects of nutrient-source reductions on water quality in the rivers and streams of the basin. The second section describes water quality characteristics of the northern Gulf of Mexico and various approaches for modeling estuarine and coastal waters. It also describes the modeling approach used to forecast the possible effects of reductions in nutrient loading from the MRB on hypoxia in the nearshore Gulf of Mexico and presents sources of data for the model simulations, spatial and temporal scales, and model calibration.

2.1 MISSISSIPPI RIVER BASIN

2.1.1 Description of the Basin and Major Sub-basins

The MRB drains 41% of the conterminous United States, including all or part of 31 states. The drainage basin is very diverse in terms of landscapes, ranging from arid areas of northeastern New Mexico and alpine regions of the eastern Rocky Mountains near its western limits, to prairie grasslands and rich farmlands in the midwestern Corn Belt and Mississippi Delta in its central region, to the industrialized region of the upper Ohio Valley, and to the heavily forested Appalachian Mountains at its southeastern limits. This vast basin discharges almost a million metric tons (0.95×10^6 Mg) per year of nitrate-N (1.57×10^6 Mg of total nitrogen) from natural and anthropogenic sources to the Gulf of Mexico (Goolsby et al. 1999). The basin also discharges approximately 137,000 metric tons (1.37×10^5 Mg) of total phosphorus to the Gulf. Partly in response to this load of nutrients, a zone of hypoxia has developed in the Gulf of Mexico. The size of the zone has varied in response to climatic variations from nearly zero in the drought years of the late 1980s to over 17,000 km^2 after the severe floods of 1993 (Rabalais et al. 1999).

The Mississippi River Basin can be divided into sub-basins in several ways, depending on one's goals and interests. For example, Task Group 3 divided the MRB into nine sub-basins, primarily on the basis of the location of hydrologic gauging and water quality sampling stations for the purpose of nutrient flux and yield estimates. Task Group 4 divided the basin into six sub-basins: the Missouri (entering the Mississippi River (MR) at Hermann, MO), the Ohio (entering the MR at Cairo, IL), the Tennessee (entering the Ohio shortly before the latter's confluence with the MR), the White/Arkansas (entering the MR below Little Rock, AR), the Upper Mississippi (the drainage basin to Cairo, IL), and the Lower Mississippi (the drainage basin below Cairo, IL). These six sub-basins are based on a U.S. Geological Survey (USGS) division of the conterminous United States into 18 water resource regions (e.g., Seaber et al. 1987).

The hydraulic and nitrate loads from the six sub-basins vary considerably. For example, according to discharge and nutrient estimates presented by Goolsby et al. (1999), the Ohio sub-basin generates 30% of the flow in the MRB and about 31% of the nitrate load to the Gulf. In contrast, the Upper Mississippi River sub-basin generates about 19% of the flow in the Mississippi River basin and 43% of the nitrate load. The Lower Mississippi contributes 13% of the water flow in the river but only 6% of the nitrate discharged to the Gulf of Mexico. The Missouri generates 13% of both the flow and the nitrate load discharged from the MRB into the Gulf of Mexico. The two other sub-basins (Tennessee and White/Arkansas) contribute small portions of the nitrate load (3% and 4%, respectively). Thus the Upper Mississippi and Ohio sub-basins generate nearly three-quarters of the total nitrate discharged from the MRB into the Gulf.

The Upper and Lower Mississippi and Ohio River sub-basins all have different climatic, soil, and land-use characteristics. Mean annual precipitation is greatest in the Lower Mississippi (> 122 cm), intermediate in the Ohio River (81–130 cm), and least in the Upper Mississippi River basin (56–100 cm), but on average all three sub-basins have abundant precipitation. Large portions of the Upper and Lower Mississippi River sub-basins, as well as areas in the northern portion of the Ohio River sub-basin, have seasonally high water tables as a result of the abundant precipitation and poor internal drainage of their soils.

Soils in the Upper Mississippi River sub-basin are dominated by Mollisols formed in glacial till and loess. These Mollisols have thick, fine-textured surface horizons, are rich in organic matter, and have poor internal drainage. The Upper Mississippi River sub-basin also has Alfisols, which have a clay-rich subsurface horizon, and surface horizons that are thinner and contain less organic matter than Mollisols. Soils in the Ohio River sub-basin include Alfisols and some Ultisols, which are highly weathered and leached by heavy precipitation, have a clay-rich horizon, and are low in soil fertility and organic matter. Soils in the Lower Mississippi River sub-basin are dominated by Alfisols and Ultisols and generally have very slow permeability.

Total cropland as a percent of total land area is greatest in the Upper Mississippi River sub-basin (mostly corn and soybeans), followed by the Ohio River (corn and soybeans in the northern half) and the Lower Mississippi River sub-basins. Total fertilizer applications follow the same ranking. In contrast, irrigation is greatest in the Lower Mississippi River sub-basin, followed by the Upper Mississippi River and Ohio River sub-basins. The five states in the Mississippi River Basin with the greatest application rates of nitrogen fertilizers (Knox and Moody 1991), the greatest fraction of cultivated land (Knox and Moody 1991), and the greatest amount of artificially drained soil (Zucker and Brown 1998) are Illinois, Indiana, Iowa, Ohio, and Minnesota.

2.1.2 Data Sources and Analytical Methods

Three primary data sources were used to evaluate the effects of nutrient-source reductions in the MRB on the quality of its rivers and streams:

- First, a large database on water flows and water quality assembled by the USGS was used for most of the basin-wide analyses. Data files of flow and of nutrient loading and concentrations at various river and stream sites were supplied to the task group by D. Gools-by, USGS, Denver, CO, chair of Task Group 3. Other data on the basin were obtained from a CD-ROM supplied by R. Alexander, USGS, Reston, VA.
- The second major source was a database on water quality, water flows, and basin landscape conditions for the Minnesota River Basin, which was assembled from a variety of agency sources by D. Mulla and co-workers at the University of Minnesota, St. Paul.
- The third source was a file of long-term data on nutrients and related water quality data for seven river sites in the metropolitan Minneapolis–St. Paul area (Upper Mississippi River Basin), which was obtained from C. Larson, Metropolitan Council Environmental Services, St. Paul, MN.

Standard statistical methods and basic spreadsheet programs were used to analyze and graph the data. Results from a variety of hydrologic/water quality models (e.g., SPARROW, EPIC, SWAT, ADAPT) also were used in various parts of the analysis. For ease in understanding the results from these models, background information on the modeling approaches is presented in Chapter 3, in conjunction with the model results.

2.2 GULF OF MEXICO

2.2.1 Site Characteristics

Seasonal hypoxia ($\leq$ 2 mg O_2/L) occurs over extensive areas (up to 18,000 km^2) in the bottom waters of the northern Gulf of Mexico inner continental shelf from May through September (Rabalais et al. 1999). The two principal factors leading to the development and maintenance of hypoxia are physical stratification of the water column and decomposition of organic material. Spatial and temporal variations in the distribution of hypoxic water masses are related, in part, to freshwater discharge from the MRB, circulation patterns, nutrient loadings, and a close coupling with net primary productivity. Significant increases in nitrogen and phosphorus loadings and decreases in silica loadings have occurred in the Mississippi River this century, and these trends have accelerated since the 1950s (Turner and Rabalais 1991). These changes appear to have caused phytoplankton species shifts and an increase in primary production offshore (Rabalais et al. 1996). Justić et al. (1993) showed that MRB inputs, net productivity, and hypoxia in the northern Gulf of Mexico are closely correlated.

Water circulation on the Louisiana–Texas shelf is strongly influenced by wind stress and freshwater discharges from the MRB (Wiseman et al. 1997; Walker 1996). About 30% of the flow from the MRB is delivered through the Atchafalaya River Delta. The remaining 70% flows through the Mississippi birdfoot delta, and eventually discharges approximately 50% to the west of the delta and 50% to the east (U.S. Army Corps of Engineers (USACE) 1984). The water flowing westward ultimately forms the Louisiana Coastal Current (LCC) (Wiseman and Kelly 1994). During much of the year, the LCC flows into Texas and Mexican waters (Cochrane and Kelly 1986). At these times, the distribution of excess fresh water is largely confined to a nearshore band that extends from the Mississippi birdfoot delta into Mexican waters (Dinnel and Wiseman 1986). Under upwelling favorable winds, which blow over the Mexican and south Texas coast from late spring through mid- or late summer, a return flow occurs (Cochrane and Kelly 1986).

The summer halocline and subhalocline thermoclines associated with the LCC isolate near-bottom waters from direct wind forcing. This effect, in conjunction with pressure gradients driving upcoast flow along the Texas inner shelf, results in slow-moving bottom waters over the LIS (Rabalais et al. 1996), allowing biological processes to deplete oxygen in the near-bottom waters. Hypoxic waters are most prevalent from late spring through late summer, and mostly in water depths of 5–30 meters (Rabalais et al. 1999). Hypoxia occurs mostly in the lower water column but encompasses as much as the lower half to two-thirds of the column.

2.2.2 Models of Estuarine and Coastal Waters

Various approaches have been developed for assessing water quality in estuarine and coastal waters. The U.S. Environmental Protection Agency (EPA) (1990) provided extensive guidance on mathematical models for assessing relationships between nutrient loading and nutrient-related water quality criteria. Hinga et al. (1995) reviewed results of three different approaches to investigate relationships between nitrogen availability and phytoplankton primary production and abundance in coastal ecosystems. These included controlled experiments in marine enclosures, assessing historical changes in coastal ecosystems, and cross-system comparisons. Wyatt (1998) investigated meteorological and anthropogenic influences on marine algal blooms and presented simple population models. Chau and Jin (1998) developed a two-layer, integrated, hydro-dynamic–eutrophication model to investigate relationships between density stratification and bottom-water anoxia in Tolo Harbour, Hong Kong.

The Water Quality Analysis Simulation Program (WASP) (Ambrose et al. 1988, 1993) is a generalized, multidimensional, mass-balance modeling framework that has been used to investigate water quality problems in a large number of different marine systems. The CE-QUAL-ICM model (Cerco and Cole 1995) is a comprehensive mass-balance model that has been used for complex problems in several large water bodies. A version of CE-QUAL-ICM was used in conjunction with a three-dimensional hydrodynamic model and a sediment diagenesis submodel to investigate eutrophication and dissolved oxygen in Chesapeake Bay (Cerco and Cole 1993; Cerco 1995a, 1995b). The Environmental Fluid Dynamics Computer

Code (EFDC) (Hamrick 1996) is a general-purpose, three-dimensional, hydrodynamic and water quality model. The EFDC model was used for studies in estuaries of Chesapeake Bay, two sites in Florida, the Peconic Bay system in New York, Stephens Passage in Alaska, and Nan Wan Bay in Taiwan.

Several recent modeling approaches involved species succession and dynamics of higher trophic levels. Roelke et al. (1997) investigated phytoplankton species succession in the Nueces River Estuary (TX) as influenced by anthropogenic activities. They evaluated and compared results from a Phytoplankton Ecology Group (PEG) model and a model based on Equilibrium Resource Competition (ERC) theory. Vasconcellos et al. (1997) used the ECOPATH approach to conduct simulations of 18 different marine trophic models to explore the behavior of systems affected at intermediate trophic levels. Manickchand–Heileman et al. (1998) developed a trophic mass-balance model to investigate energy flow in a community of fish and invertebrates in the southwestern portion of the Gulf of Mexico.

2.2.3 Methods

2.2.3.1 MODELING APPROACH

The use of mathematical models for investigating hypoxia in the northern Gulf of Mexico is at an early stage of development. As part of the NOAA Nutrient Enhanced Coastal Ocean Productivity (NECOP) program, Bierman et al. (1994a) applied a version of the EPA WASP model to the LIS portion of the northern Gulf of Mexico (Figure 2-1). The development of this model was a collaborative effort among Limno-Tech, Inc. (LTI), Louisiana State University (LSU), and the Louisiana Universities Marine Consortium (LUMCON). In addition, the modeling effort drew extensively from field monitoring and research conducted by other investigators in the NECOP program.

The NECOP program was not designed to collect field monitoring data to support a mass-balance water quality model. Modeling was only one of several complementary, parallel, program

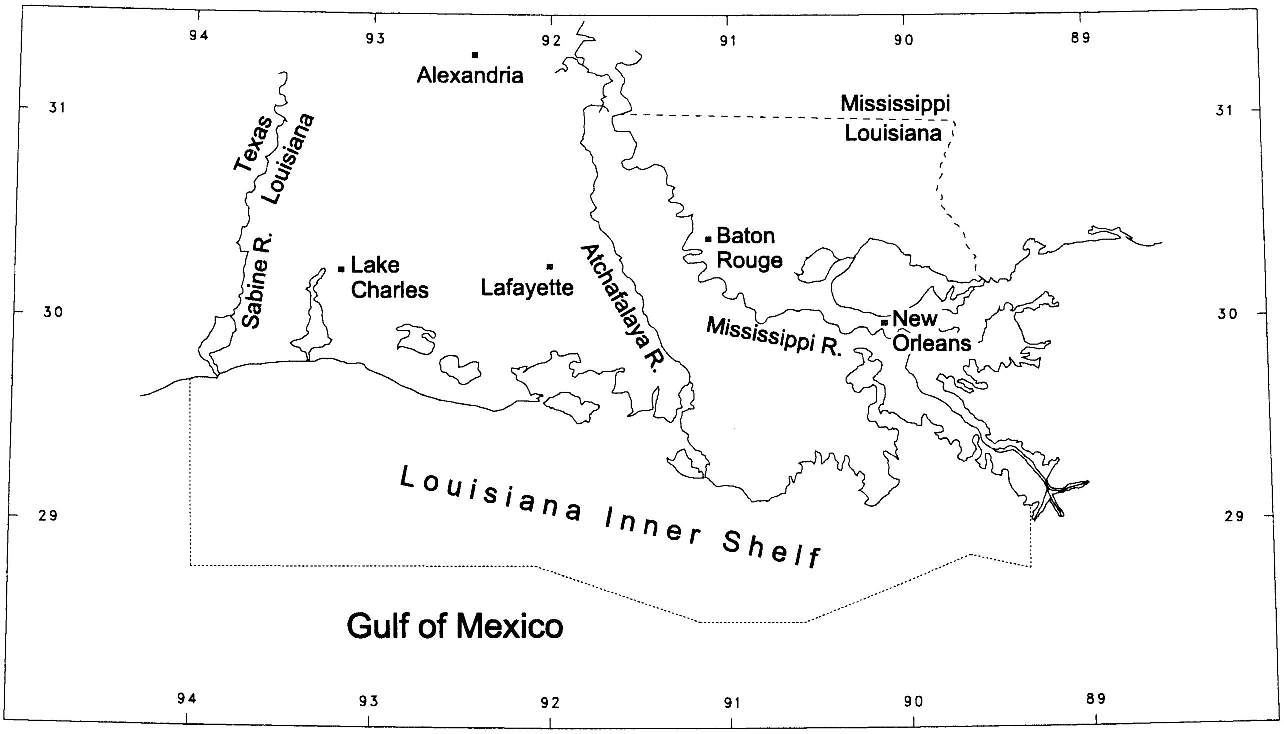


FIGURE 2.1. Location map of the study area for the water quality model.

elements. Available historical data and new data generated within the NECOP program were sufficient for only a preliminary, screening-level modeling analysis. Until the NECOP modeling study, there had been no previous applications of mass-balance water quality models on the LIS. Although there are many uncertainties in model results, the NECOP model was used to address broad, macro-scale questions related to the impacts of potential reductions in nutrient loadings from the MAR. The Gulf of Mexico sections of this report describe the NECOP model and results from forecast simulations designed to estimate responses of dissolved oxygen and chlorophyll concentrations to potential reductions in nitrogen and phosphorus loadings from the MAR.

The conceptual framework for the modeling approach is shown in Figure 2.2. State variables in the model include salinity, phytoplankton carbon, phosphorus (dissolved orthophosphate and combined P forms), nitrogen (ammonium, nitrate plus nitrite, and organic N forms), dissolved oxygen, and carbonaceous biochemical oxygen demand (CBOD). User-specified, external forcing functions include constituent mass loadings, advective–dispersive transport, seaward boundary conditions, sediment fluxes, water temperature, incident solar radiation, and underwater light attenuation. Sediment interactions are represented by user-specified values for net settling rates for particulate phase constituents, sediment–water diffusive fluxes for dissolved nutrients, and sediment oxygen demand (SOD). This conceptual model was implemented using an LTI-modified version of the EPA WASP computer modeling framework.

The model represents the complete mass-balance cycle for each state variable in the water column. This cycle includes mass inputs, outputs, and transformations for each state variable as a function of space and time. With respect to nutrients, the model represents forms that are immediately available for phytoplankton growth as well as forms that are not immediately available but can become available through mineralization in the water column. The model represents available nitrogen as the sum of ammonium and nitrate plus nitrite forms, and available phosphorus as dissolved orthophosphate. Unavailable nitrogen and phosphorus are lumped into the state variables organic nitrogen and organic phosphorus, respectively. Mineralization of organic nutrient forms to available nutrient forms is represented in the model by first-order, temperature-dependent mechanisms. Consequently, nutrients for phytoplankton growth are supplied not only from external mass inputs but also from internal mineralization in the water column.

The WASP model framework was selected for two principal reasons: (1) it contains only a moderate degree of complexity and was reasonably compatible with the available field data, and (2) it could provide first-order answers to the principal water quality questions. Although this model contains only a moderate degree of chemical–biological complexity, it still requires a considerable amount of field data for specification of external model-forcing functions, as well as for comparison with model output.

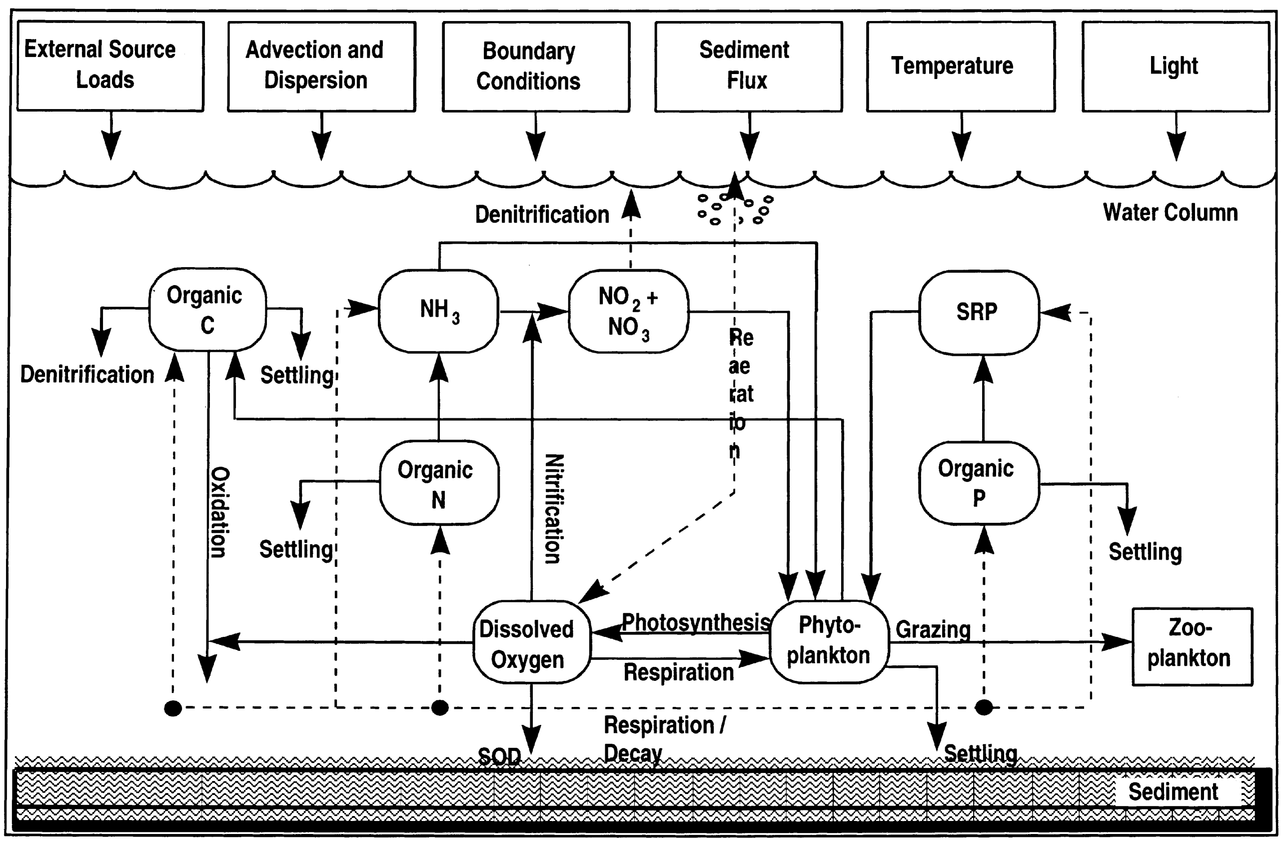


FIGURE 2.2. Schematic map of the principal model state variables and processes in the Gulf of Mexico water quality model.

2.2.3.2 DATA SOURCES

The principal application data were drawn from a comprehensive set of field studies conducted during July 1990 at over 200 sampling stations in the northern Gulf of Mexico (Figure 2.3).

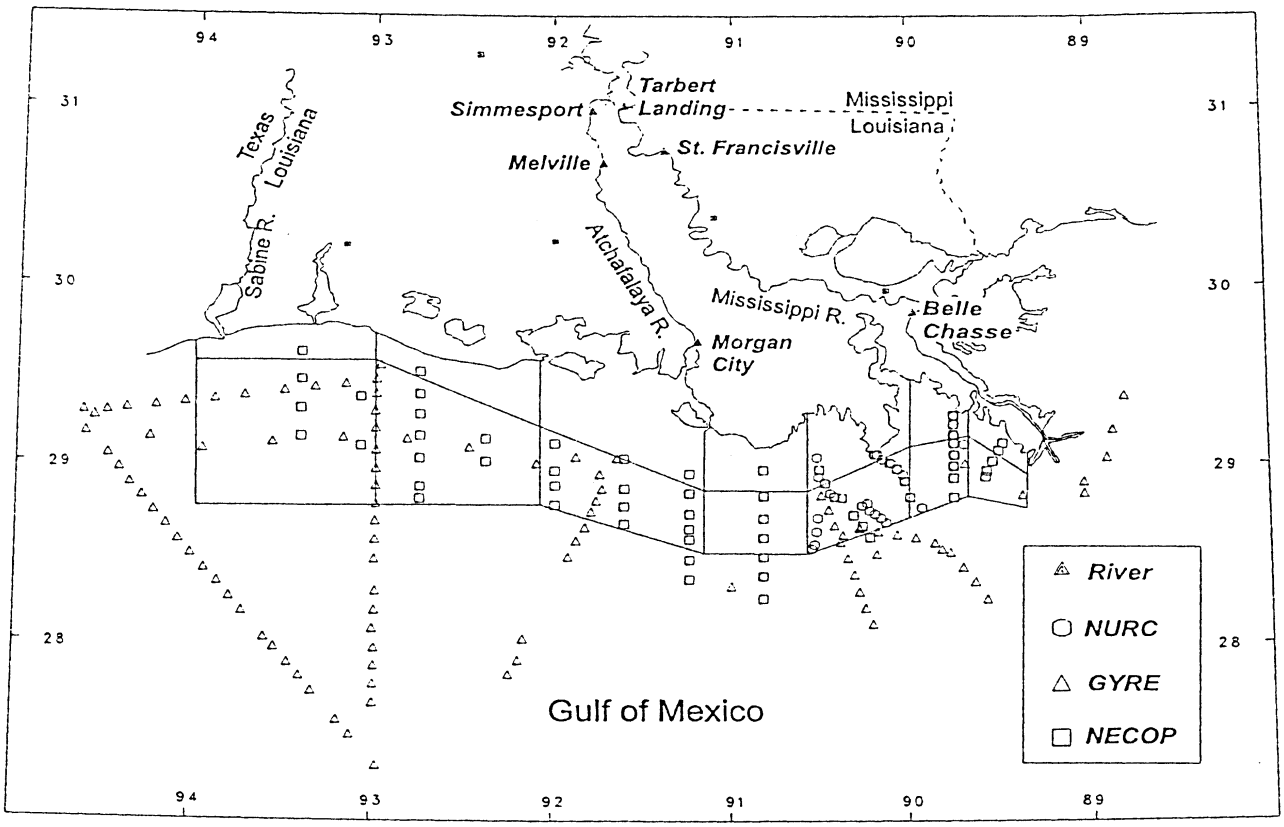


FIGURE 2.3. Location of the sampling stations used in model calibrations for July 1990.

Four groups of sampling stations were included in these studies: the NECOP–NECOP90 shelfwide cruise, which occupied 64 stations located primarily inside the model's spatial domain; the GYRE–GYRE90 cruise conducted by Texas A&M University, which occupied 113 stations located both inside and outside the model's spatial domain; the NURC–NURC90 cruise conducted by Louisiana Universities Marine Consortium, Texas A&M University at Galveston, and Texas Institute of Oceanography, which occupied 38 stations located immediately west of the Mississippi Delta in the primary hypoxic region; and the River–USGS stations in the Mississippi and Atchafalaya Rivers. The field data from all these sampling stations (except river stations) reside in the NECOP database management system (Hendee 1994). The model was also applied to earlier historical data collected during 1985–88 by LUMCON (Rabalais et al. 1996).

2.2.3.3 SPATIAL AND TEMPORAL SCALES

The spatial domain of the NECOP model is represented by a 21-segment water-column grid extending from the Mississippi River Delta west to the Louisiana–Texas border, and from the shoreline seaward to the 30–60 m bathymetric contours (Figure 2.4). The spatial segmentation grid includes one vertical layer nearshore and two vertical layers offshore. All spatial segments are assumed to be completely mixed. The nearshore segments have an average depth of 5.6 m. The surface offshore segments are completely mixed in the vertical to an assumed fixed pycnocline depth of 10 m. The bottom offshore segments are completely mixed from 10 m to the seabed. The depths of these bottom offshore segments range between 6.1 and 20.3 m.

The coarse scale of the model's segmentation grid was originally determined by the areal distribution and vertical density of historical water quality data. Salinity was used to identify characteristic water masses and to determine the geometric boundaries of the grid. The scale of the model's segmentation grid has two principal limitations: (1) near-field horizontal gradients in the vicinity of the Mississippi and Atchafalaya River plumes are not explicitly represented; and (2) important vertical-scale characteristics are not fully represented, including near-bottom hypoxia and "layering" of dissolved oxygen concentrations. Rabalais et al. (1999) showed that the low-oxygen water mass in bottom waters can move and change configuration in response to winds, currents, and tidal advection. Furthermore, they showed that hypoxia can occur not only at the bottom (near the sediments), but also well up into the water column.

The temporal domain of this model application represents steady-state, summer-average conditions. Consequently, the model represents only a single "snapshot" in time. In reality, there is great daily and weekly variability in current flow and stratification on the LIS (Rabalais et al. 1996). The principal reason for this model limitation is that field measurements are not available to characterize temporal variability at the shelfwide spatial scale. Typically, only a single shelfwide monitoring effort is conducted each year during the July–August period to characterize the spatial extent of hypoxia. Operationally, model-forcing functions were assigned constant values that represented summer-average conditions. The time-variable model was then run to steady-state, and model output was compared with available field data. It was assumed that data collected during the summer shelfwide monitoring effort were synoptic, and that they were in temporal equilibrium with the specified summer-average model-forcing functions.

2.2.3.4 MODEL-FORCING FUNCTIONS

Physical transport in the model is represented by advective flow and bulk dispersion. Bulk dispersion is a lumped parameter that represents transport processes at scales smaller than the model's spatial segments. These processes include molecular diffusion, turbulent eddy diffusion, and shear-flow dispersion. Because the model balances mass and not momentum, magnitudes and directions for advective flows must be externally specified by the user. Dispersive mixing coefficients across all horizontal and vertical interfaces are calibration parameters and were determined by conducting mass balances for salinity, a conservative tracer.

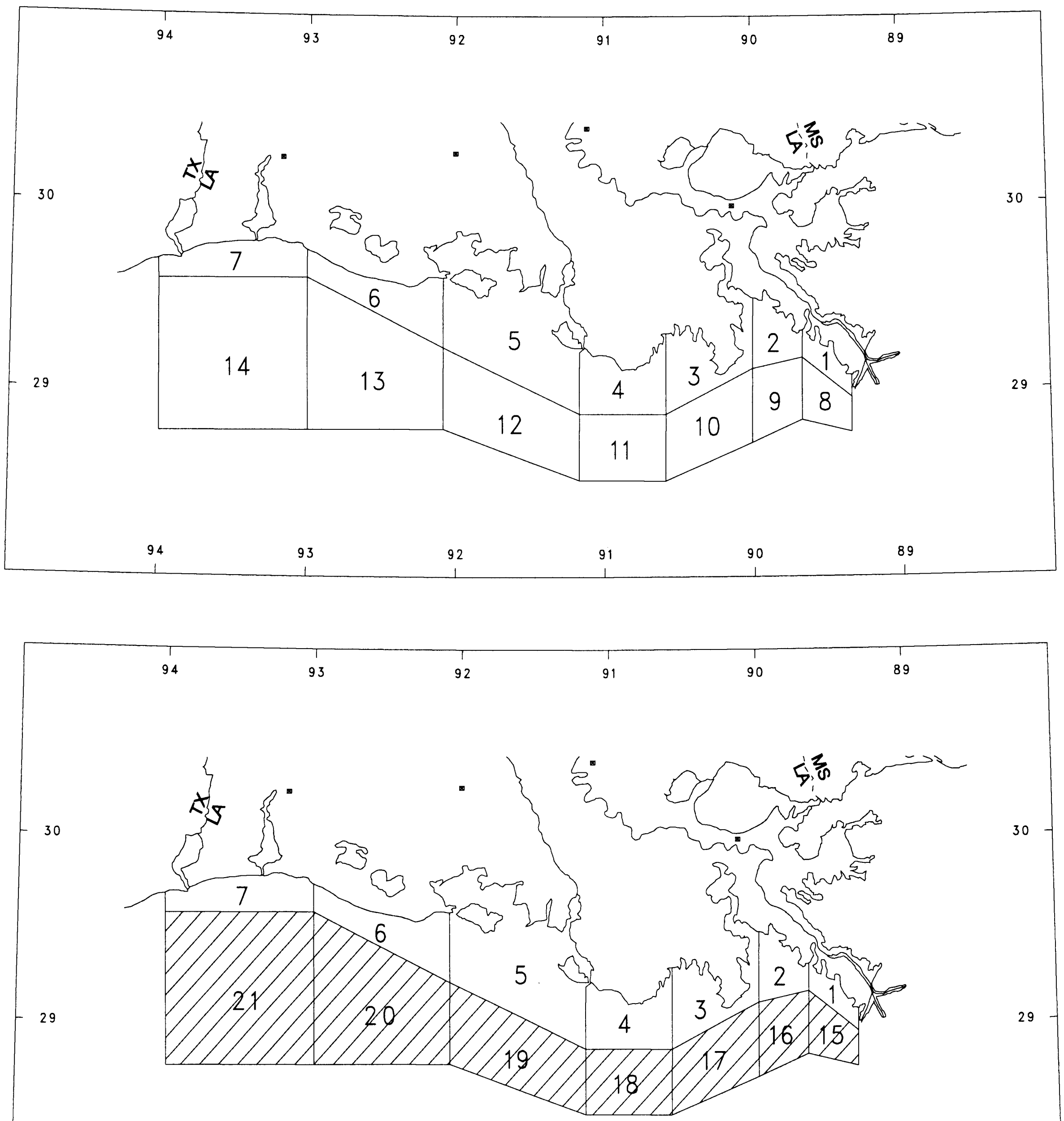


FIGURE 2.4. Model spatial segmentation grid for the Louisiana Inner Shelf portion of the Gulf of Mexico. NOTE: Segments 15–21 are the bottom-water segments underlying surface segments 8–14.

Advective flows in the model are descriptive in nature and were based on relatively sparse observational data. It is believed that summer-average conditions in the spatial domain of the model are typically represented by the LCC, which has a net westward drift along the shelf bathymetry. This representation is supported by current measurements from a long-term mooring maintained by W.J. Wiseman, Jr., LSU, at a location off Cocodrie (segment 10) in 20 m of water. Typical summer-average current speeds are ~10 cm s^{-1} and ~3 cm s^{-1}, respectively, in the surface and bottom waters.

Although the model was calibrated using the most comprehensive data set available at the time, a question arose as to whether this particular calibration was representative because hydrometeorological conditions on the LIS appeared to be anomalous during the summer of 1990. In contrast to results from the above long-term current meter record, net eastward drift was observed in both surface and bottom waters at speeds of approximately 2 cm s^{-1} and 0.8 cm s^{-1}, respectively. Consequently, before using the calibrated model to conduct forecast simulations, it was deemed appropriate to calibrate the model to a wider range of environmental conditions. Prior to conducting the forecast simulations in the present report, the model calibration was extended to include summer-average conditions in July 1985 and August 1988. Selection of the July 1985 and August 1988 data sets was based on differences among individual years with respect to numbers of sampling stations occupied, magnitudes of MRB inflows, and areas of hypoxia.

Available information indicates that a typical LCC existed during the summers of 1985 and 1988. Annual average inflows from the MRB in 1985 (845,000 cfs) and 1990 (877,000 cfs) were higher than the long-term (1930–92) annual average inflow (664,000 cfs). Peak flows generally occur in April, although peak inflow in 1990 occurred in June. Annual average inflow in 1988 (535,000 cfs) was lower than the long-term average inflow, and historical low flows occurred during the summer of 1988. Areas of hypoxia were much greater in the summers of 1985 and 1990 (approximately 9,000 km^2) than in the summer of 1988 (approximately 40 km^2) (Rabalais et al. 1999). Although including data sets for July 1985 and August 1988 extends the range of model calibration conditions, field data for these periods are not nearly as comprehensive as field data for July 1990, in terms of both numbers of stations occupied and numbers of water quality parameters measured.

To the extent permitted by available field data, external forcing functions for the model calibration data sets were year-specific. The most important differences among the three summer-average calibration periods were differences in MRB inflows and freshwater advective flow magnitudes and directions on the LIS (Figures 2.5–2.7). These freshwater advective flow fields represent our best judgment in synthesizing available field data for riverine discharges, observed current speeds and directions, and satellite imagery. Dispersive mixing coefficients across all horizontal and vertical interfaces were determined by conducting mass balances for salinity, a conservative tracer.

Table 2.1 summarizes the tributary inflows and nutrient loadings for the three model application periods. These forcing functions represent average MRB conditions for antecedent periods of approximately one month for each of the three calibration data sets. Inflow to the model grid from the Mississippi River was much lower than inflow from the Atchafalaya River during the July 1990 application. This was necessary to match observed salinities in the model segments near the delta and was likely due to the net eastward drift in water circulation during this period.

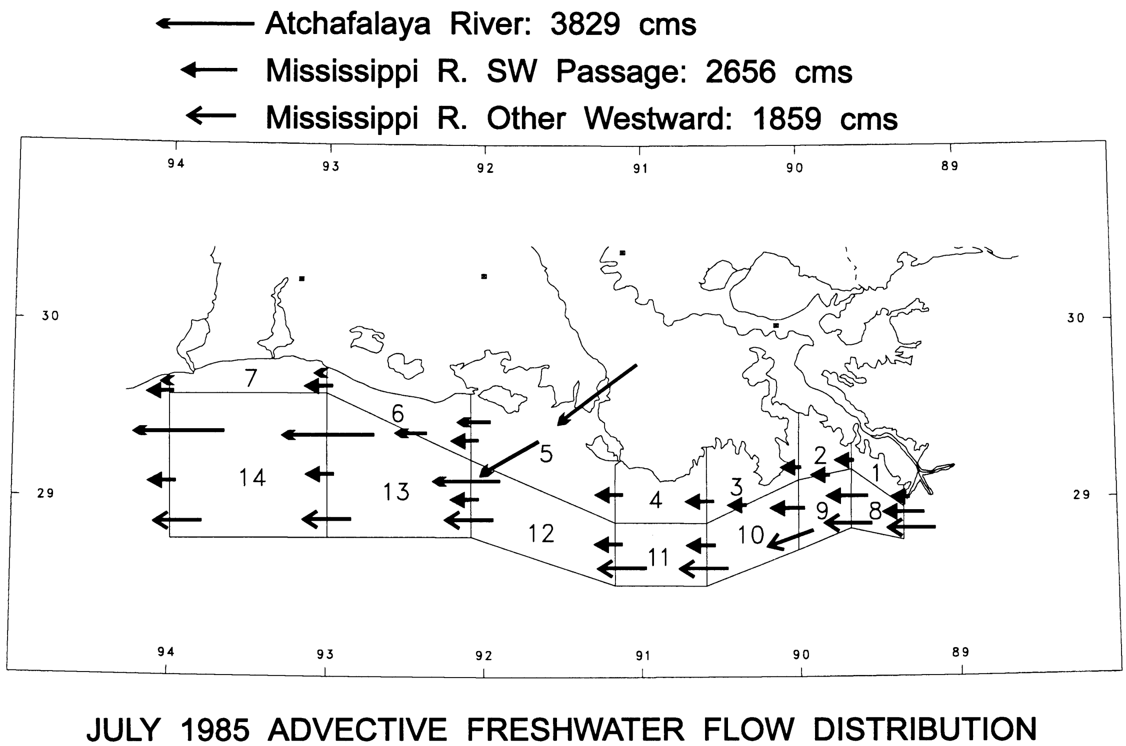


FIGURE 2.5. Schematic diagram of freshwater advective flows used in model calibrations for July 1985.

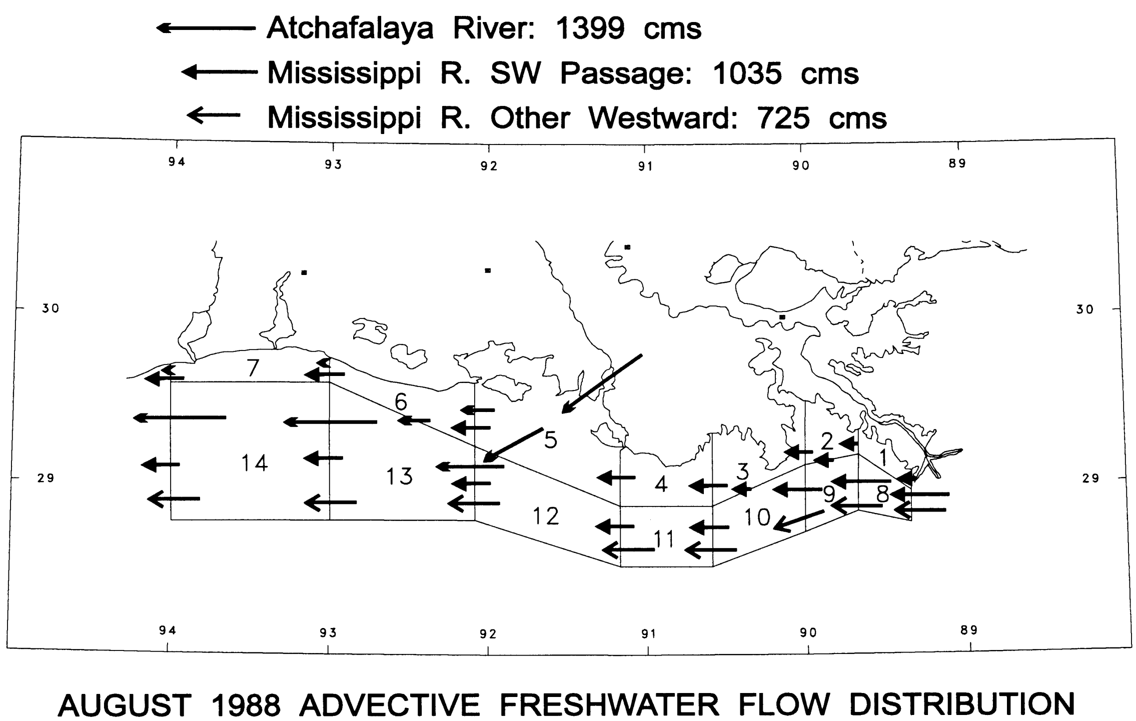
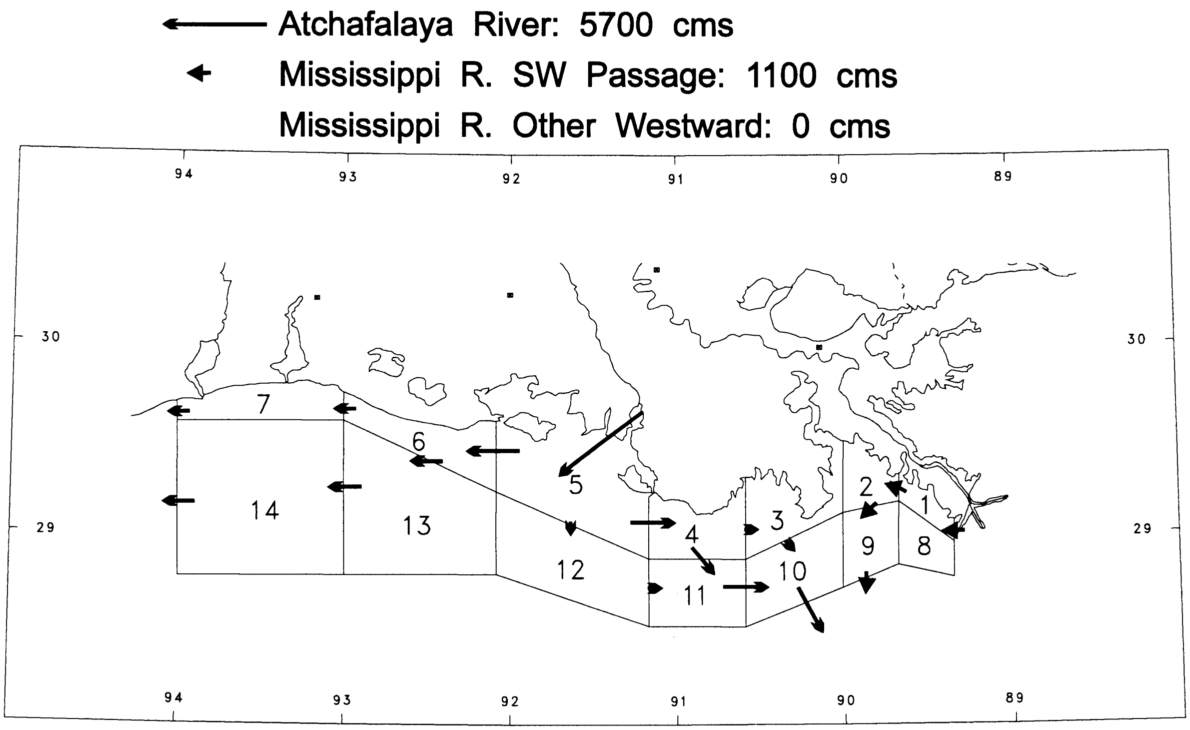


FIGURE 2.6. Schematic diagram of freshwater advective flows used in model calibrations for August 1988.



JULY 1990 ADVECTIVE FRESHWATER FLOW DISTRIBUTION

FIGURE 2.7. Schematic diagram of freshwater advective flows used in model calibrations for July 1990.

TABLE 2.1. Tributary inflows and nutrient loads in base calibration.

Parameter		**Mississippi**			**Atchafalaya**	
	1985	***1988***	***1990***	***1985***	***1988***	***1990***
Inflow[1] ($m^3 s^{-1}$)	4,515	1,760	1,100	3,829	1,399	5,700
N Load (metric tons/day^{-1})	956.0	149.40	294.0	595	114.8	911
Inorganic[2]	*671.0*	*116.00*	*230.0*	*324*	*62.8*	*458*
Organic N	*285.0*	*33.40*	*64.2*	*271*	*52.0*	*453*
P Load (metric tons/day^{-1})	78.0	15.20	28.50	33.0	9.07	98.6
Available[3]	*46.8*	*9.12*	*5.70*	*19.8*	*3.63*	*29.6*
Unavailable[4]	*31.2*	*6.08*	*22.80*	*13.2*	*5.44*	*69.0*

[1]*Sum of Mississippi inflows from Southwest Pass and westward flows from other passages.*
[2]*Sum of ammonium-N and nitrate plus nitrite-N.*
[3]*Dissolved orthophosphorus.*
[4]*Total phosphorus minus dissolved orthophosphorus.*

2.2.3.5 MODEL CALIBRATION

Detailed calibration results using the comprehensive July 1990 field data are presented in Bierman et al. (1994a). Reasonable comparisons were obtained between computed and observed values for model state variables, primary productivity, and mass settling fluxes for particulate carbon and nitrogen. For the present analysis, chemical–biological state variables were calibrated using internal model parameters from the July 1990 calibration as starting values, and were then adjusted to obtain optimal results across all three summer-average periods. The final calibration consisted of a unified set of internal model parameters that produced the best average results for the three summer periods. Calibration results for any individual summer period do not necessarily represent the best results possible for that period. This calibration approach was based on the judgment that differences among summer-average periods were due primarily to differences in environmental forcing functions, not internal model processes.

Detailed results for this unified model calibration are presented in Limno-Tech, Inc. (1995). As a gross quantitative measure of goodness-of-fit, overall mean values for model output and field data were compared for each parameter-year combination using the Student's "t" test. These overall mean values represented grand averages of individual segment mean values. There were 18 parameter-year combinations, and model output was significantly different ($p < 0.05$) from field data in only three of the 18 cases. Regression analyses of model output versus observed segment mean values were also conducted for the 18 cases. Results indicated that although the model represented the overall mean state of the system reasonably well, it explained an average of only 40% of the spatial variability among individual model segments. This is probably due to the fact that a complex, dynamic system is being represented as a single "snapshot" in time and at coarse spatial scales. Rabalais et al. (1996) emphasized that there is great daily and weekly variability in current flow and stratification on the shelf and that there is no simple description of the important physical–biological couplings.

CHAPTER 3

Results

3.1 INTRODUCTION

Nutrient-source reductions in the Mississippi River Basin (MRB) may be implemented with the primary goal of decreasing the hypoxia problem in the Gulf of Mexico, but they also will affect water quality conditions in the MRB itself. The nature of the responses to such reductions is quite different in the two systems. In the Gulf, the response variable of primary interest is dissolved oxygen (in the bottom waters); in freshwater parts of the MRB, responses of a much wider array of water quality variables—various nutrient forms, chlorophyll, dissolved oxygen, water clarity, planktonic and benthic biota—are of direct interest.

The methods available to analyze the effects of nutrient-source reductions also differ significantly for the two systems. For the Gulf of Mexico, a dynamic simulation model is available to make quantitative predictions on the effects of changes in nutrient loads on dissolved oxygen concentrations. At present, no model is available to simulate the entire MRB freshwater system, although many models can be used to examine specific aspects of the nutrient loading–aquatic response issue for small parts of the system.

Hypoxia in the Gulf is thought to be driven by the loading of nitrogen (N) primarily as nitrate from the MRB because nitrogen generally is considered to be the limiting nutrient for phytoplankton production, and phosphorus is thought to be present in excess of plant needs in marine waters. Therefore, our analysis of the effects of nutrient loading reductions from the MRB on hypoxia in the nearshore Gulf focuses on N loadings.

In contrast, aquatic scientists regard phosphorus (P) as the nutrient that most frequently limits plant growth in freshwater ecosystems. Even in freshwater situations where phosphorus is present in excess (usually because of excess inputs from human activities), aquatic scientists traditionally have focused on the effects of controlling phosphorus under the assumptions that (1) P loadings are more readily controlled than are N loadings, and (2) phosphorus can be made the limiting nutrient again by sufficiently reducing its inputs.

In recent years aquatic scientists have recognized that the ratio of nitrogen to phosphorus is important in controlling the composition of phytoplankton species and that nitrogen in fact may be the limiting nutrient in many freshwater systems. Consequently, efforts to control eutrophication in freshwater systems have taken a more balanced approach that considers loadings of both N and P and attempts to control sources of both. Our analysis of the effects of nutrient-source reductions on water quality in the freshwater ecosystems of the Mississippi River Basin thus considers both N and P.

Because of these differences, our analysis of aquatic system responses to nutrient-source reductions is presented in separate parts. Section 3.2 examines the likely effects of such reductions on nutrient concentrations and loads in the river system and then considers the effects of changes in nutrient concentrations on a variety of water quality characteristics, including algal biomass and productivity. Section 3.3 examines response of dissolved oxygen in bottom waters of the nearshore Gulf of Mexico to a range of reductions in N loadings from the Mississippi River.

3.2 MISSISSIPPI RIVER BASIN

3.2.1 Categories of Nutrient Sources

From the perspective of this report, nutrient sources to the Mississippi River and its tributaries can be divided conveniently into two categories: those that provide nutrients directly to the water courses themselves, and those that provide nutrients to the terrestrial landscape, from which the nutrients then are transported into water bodies. In practical terms, the former category is identified primarily as point-source discharges—municipal and industrial wastewater effluents—and the latter primarily as nonpoint, or diffuse sources—agricultural and urban runoff, atmospheric precipitation. Of course, some diffuse sources of nutrients contribute directly to water bodies as well as to terrestrial landscapes—rainfall occurs on rivers and lakes as well as land—but the direct contributions to these freshwater systems are small compared with the contributions to the terrestrial part of the drainage basin. To the extent that stormwater runoff from urban areas is collected and transported to water courses by storm sewer systems, this source may be considered a point-source discharge. Typically, however, many storm sewer outfalls are spread over a given urban area, and urban runoff thus can be considered a diffuse source of nutrients. A similar situation exists for agricultural stormwater in many areas where artificial drainage (tile drains) has been installed to accelerate the removal of rainfall from otherwise poorly drained soils.

The distinction between direct and indirect sources of nutrients to the river is important relative to analyzing the effects of source control measures on concentrations and loads of nutrients in the Mississippi River. Controls that are implemented on direct sources will have a direct and proportionate effect on loads in the river. That is, if municipal wastewater plants decreased the N content of their effluents by 50%, their N mass contributions to the river system also would decrease by 50%. (It cannot be stated that the mass load of N from the Mississippi River to the Gulf of Mexico would decrease by the same amount because, as described elsewhere in this report, there is some retention and loss of N within the riverine system.) In contrast, if the mass of N fertilizer applied to fields were decreased by a specific quantity in the Corn Belt, we could not predict that N discharges to the river and its tributaries from agricultural areas of the Corn Belt would decline by that amount, because fertilizer use is only one factor (albeit an important one) affecting the loss of nitrogen from agricultural lands. Indeed, because of nonlinearities and compensating mechanisms in nutrient cycling and transport processes, it is impossible even to predict that the same percentage reductions will occur in river loads of nitrogen from diffuse sources for a given percentage reduction in inputs of nitrogen to the MRB. Some exceptions to this situation are described later in this report, but two important general considerations should be kept in mind regarding diffuse sources of nutrients in the MRB:

- Our ability to predict the effects of reductions in nutrient inputs to the basin on nutrient export to the river and on nutrient concentrations and loads in the river is limited.
- However, because of known compensating mechanisms and nonlinearities in nutrient cycling and transport processes, the (mass) decrease in nutrient export to the Gulf by the Mississippi probably will be less than any aggregate (mass) decrease in nutrient inputs to the MRB.

3.2.2 Relative Importance of Nutrient Categories

In terms of the major categories of nutrient sources within the MRB, as described above, Goolsby et al. (1999) concluded in the Task Group 3 report that approximately 10% of the nitrogen in the Mississippi is from point-source discharges (primarily municipal wastewater treatment plants), 10% from atmospheric deposition, and 80% from diffuse (nonpoint) sources.

The nitrogen in atmospheric deposition consists of nitrate, ammonium, and organic N forms. Each is derived from a different source. Most organic N comes from soil and plant material via wind-blown suspension of particles into the atmosphere. Because the scale of atmospheric transport of such particles usually is relatively short, most organic N in atmospheric deposition represents recycled N from the watershed where the deposition is occurring. (This statement also applies to all the P forms found in atmospheric deposition.) For large drainage basins like the MRB, essentially all the organic N and TP load from atmospheric deposition falls into this category. The ammonium in atmospheric deposition is derived primarily from the volatilization of two agricultural sources: (1) ammonia-based fertilizers and (2) ammonia (NH_3) mineralized from manure in feedlots and other confined animal operations. For the most part, this also represents recycled rather than new N for large watersheds. The nitrate in atmospheric deposition is derived primarily from conversion of NO_x to nitric acid in the atmosphere. The NO_x is emitted to the atmosphere as a result of fossil fuel combustion (gasoline and diesel fuel in cars and trucks, coal and oil in power plants); this represents an additional source of nitrogen to watersheds.

Although Goolsby et al. (1999) were unable to separate the urban and agricultural contributions from the diffuse sources, several lines of evidence indicate that urban contributions are much smaller than rural/agricultural contributions.

- Urban areas occupy only a very small portion (< 1%) of the MRB, and agricultural areas occupy a much larger fraction (~58%, not including grazed lands).
- Although the ranges of N and P concentrations in urban and agricultural runoff are large, the ranges for a given nutrient generally overlap. As a first approximation, we may consider the average concentrations to be the same.
- Areal export rates of N and P from urban and agricultural areas generally have similar (and large) ranges.

In some cases, nutrient export coefficients for urban areas are somewhat larger than those for agricultural areas, mostly because of the larger fraction of impervious surfaces and, hence, higher quantities of runoff in urban areas (Novotny and Olem 1994; Carpenter et al. 1998). However, the differences are not consistent, nor are they sufficient to compensate for the large difference in area between these two source categories. Therefore, as a rough approximation, we estimate that urban runoff contributes only a small percentage of the total diffuse-source loading to the MRB, and runoff from agricultural lands contributes most of the remaining loading.

Areal export rates of nutrients from forests and grasslands are quite small compared to those from urban and agricultural lands. Thus, the contributions of these lands to the total nutrient load of the MRB are relatively small, in spite of the large area of the MRB they occupy. This conclusion is supported by the relatively low values of nutrient loadings from sub-basins in which forests and grasslands predominate (e.g., the Upper Missouri sub-basin, Figure 3.1 (Goolsby et al. 1999)).

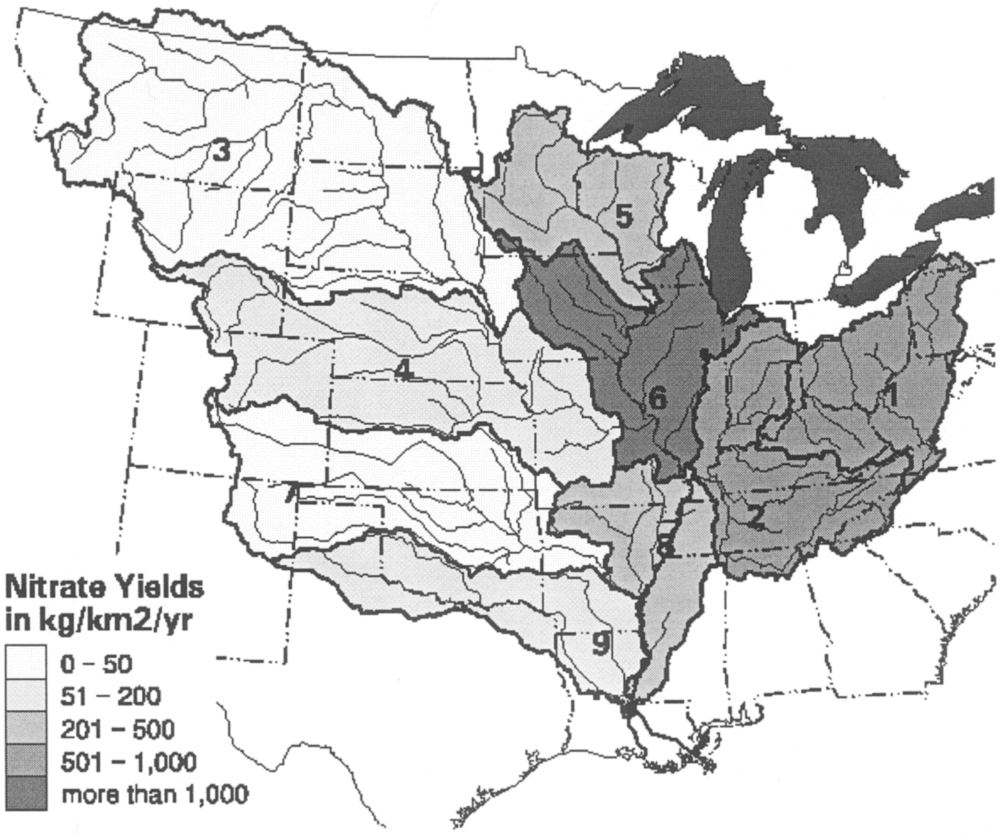


FIGURE 3.1. Contributions of major MRB sub-basins to the total nitrate load to the Gulf of Mexico. NOTE: 1 = Upper Ohio; 2 = Lower Ohio; 3 = Upper Missouri; 4 = Lower Missouri; 5 = Upper Mississippi; 6 = Middle Mississippi; 7 = Arkansas; 8 = Lower Mississippi; 9 = Red–Ouachita. *(From Goolsby et al. 1999.)*

3.2.3 Potential Strategies for Reducing Nutrient Losses from Nonagricultural Sources

3.2.3.1 REMOVING PHOSPHORUS FROM WASTEWATER

The technology for removing phosphorus from wastewater is well advanced. Many municipal treatment plants have practiced P removal for 10–20 years. For example, phosphorus removal is required for treatment plants that discharge into the Great Lakes or their tributaries. However, it is generally not required of (or practiced by) municipal treatment plants in the MRB, although at least some states (e.g., Minnesota and Wisconsin) require it for plants that discharge their effluents directly into lakes or into tributaries that drain into lakes within some prescribed distance.

Phosphorus can be removed from wastewater by chemical precipitation or microbial uptake. The former process uses calcium, iron, or aluminum salts, the selection of which depends primarily on availability and economics. Chemical removal methods generally are more expensive than biological removal, but they are more effective in reducing effluent concentrations. Removal rates of 80–90% and effluent TP concentrations of < 1 mg/L are easily achievable; under optimal conditions, effluents with TP concentrations as low as 0.1 mg/L can be produced. Biological removal methods rely on microbial uptake of inorganic phosphate in "luxury" amounts—i.e., amounts greater than the cells require for immediate growth. Many bacteria and algae take up phosphate in luxury amounts and store it in cells primarily as polyphosphate reservoirs. The key to practical use of this phenomenon in wastewater treatment plants is to design the system to promote retention of the luxury phosphate within the microbial biomass (i.e., biosolids, or "sludge"). The advantage of biological P removal is that it is relatively low cost. It requires fewer capital investments and often can be initiated in a wastewater treatment plant by changing operating practices without requiring additional facilities or hardware. In general, biological treatment does not remove phosphorus to levels as low as can be achieved by chemical precipitation, but reductions in effluent concentrations to the 1 mg/L range are achievable with this approach.

3.2.3.2 REMOVING NITROGEN FROM WASTEWATER

The technology for removing nitrogen from wastewater effluent is less advanced. Although many processes have the potential for N removal, most have been found to be ineffective, impractical, or cost-ineffective. Compared with P removal, relatively few municipal wastewater treatment plants in the United States currently practice N removal (plants in the Chesapeake Bay watershed are exceptions). Plants that do practice N removal use biological processes: a combination of nitrification (which requires oxygen) to convert the nitrogen in the wastewater to nitrate, followed by denitrification—i.e. the microbial reduction of nitrate to N_2 (which requires an absence of oxygen). Different microbial communities ("activated sludges") must be maintained to carry out this combined process efficiently. Consequently, plants using it need to enhance their treatment process facilities and are more difficult to operate. Well-run systems can achieve 80–90% removal of TN from wastewater effluents, but the costs of doing this are significant.

Most of the N and P removed from wastewater by conventional treatment plants winds up in biosolids, and a significant fraction of the biosolids produced nationally—about 20–40%—is disposed of by land application (*Federal Register* 55(218):47218, 47267). This land disposal may be another, albeit relatively small, component of nonpoint-source nutrients.

3.2.3.3 REDUCING NUTRIENTS WITH BIOLOGICAL TECHNIQUES

Some biological techniques offer the possibility of simultaneously removing N and P from wastewater effluent. Use of ponds containing macrophytes and natural or constructed wetlands as tertiary treatment systems falls in this category. Such systems work better in warmer climates than in areas with long, harsh winters. They also are more practical for treating effluents from small communities than large cities, because relatively large wetlands or pond areas are required per unit volume of effluent.

The long-term effectiveness of such "semi-natural" systems for nutrient removal is controversial and still poorly understood. For N removal, such systems could work effectively on an indefinite basis if the primary removal mechanism is denitrification. In contrast, if the primary removal mechanism is deposition of particulate organic forms, then eventually the pond or wetland will fill up or become saturated with nitrogen.

For phosphorus, the primary removal mechanisms are biological uptake and sorption onto sediments. Saturation of the system is a distinct possibility for both processes, and when this occurs, the pond or wetland is no longer effective in P removal. Concern about P saturation applies also to the use of ponds and wetlands for treatment of agricultural and urban stormwater runoff. The effectiveness of using ponds and constructed wetlands could be improved and extended by periodic harvests of the plants they produce.

3.2.3.4 REDUCING NUTRIENTS BY TREATING URBAN STORMWATER

The techniques for treating urban stormwater to remove nutrients are not highly advanced. Most of them rely primarily on slowing down the flow rate of the water, thus allowing suspended matter to settle out; stormwater retention and detention ponds fall in this category, as do grassy swales. Nutrients associated with suspended material are removed to the extent that the settling process is effective, but concentrations of dissolved nutrients are not affected much.

Frequent sweeping or vacuuming of streets to remove litter, plant debris, and soil particles also is directed at particulate, rather than soluble, nutrient forms. Overall, these methods remove only small fractions of the nutrients from urban stormwater.

Overuse of P fertilizer on lawns is especially common. Public education regarding proper use of fertilizers for lawns and gardens is widely promoted as a means for reducing P losses to urban runoff.

3.2.3.5 REDUCING NUTRIENTS FROM ATMOSPHERIC DEPOSITION

The only techniques available for reducing nutrient contributions from atmospheric deposition involve reduction of emission sources; treatment of rainfall is not a practical option. Improvements in fertilizer application practices and in manure management could reduce volatilization losses of NH_3 from these sources, and there are economic and other incentives for doing this. As pointed out in Section 3.2.2, much of the ammonia in atmospheric deposition does not represent a new source of nitrogen to the MRB, but is instead a recycling and redistribution of ammonia that already is in the basin.

3.2.3.6 REDUCING THE FORMATION OF NO_X

Technologies are not readily available, beyond those currently in use, to significantly reduce the formation of NO_x in common types of stationary and mobile sources or to achieve significantly greater removal of NO_x from these sources before it is emitted to the atmosphere. Further use of alternative types of power generation, such as nuclear or hydropower, that have intrinsically low emissions of NO_x involves other serious environmental problems. The best prospects for significant reductions in NO_x emissions from automobiles involves electric or hybrid (electric/gasoline) power trains, but these are some years away from being economically competitive with current technology. Significant additional reductions in emissions of NO_x from stationary and mobile sources beyond those currently mandated thus would be difficult and expensive to achieve.

Given that the rates of nutrient deposition onto landscapes from the atmosphere are not high enough to cause nutrient overenrichment problems in the MRB by themselves, it may be more practical to consider this deposition as a “free good” that benefits plant growth in forests and other nonagricultural landscapes. It could be accounted for it in calculating fertilizer needs for cultivated cropland, but at present, atmospheric contributions of nutrients are small compared with agronomic requirements.

On the other hand, high rates of atmospheric deposition of nitrate are associated with the phenomenon of “acid rain,” and on a national basis there are reasons to further limit NO_x emissions to control this problem, as well as other air quality problems (smog, ozone) associated with NO_x emissions. In general, acid deposition is not a problem in the MRB, except in the headwaters area of the Wisconsin River, which drains into the Upper Mississippi River, and in highland areas of the Appalachians. In both areas, the problem is associated more with sulfate than with nitrate deposition.

3.2.4 Potential Strategies for Reducing Nutrient Losses from Cultivated Land

3.2.4.1 PROCESSES AFFECTING NITROGEN FATE AND TRANSPORT

Fertilizer or manure is applied to cultivated cropland in order to provide nitrogen, the essential nutrient element for crop growth. Without adequate nitrogen, crop production and the economic viability of farming both suffer greatly. Nitrogen also can be added to the soil from the atmosphere by biological nitrogen fixation and by mineralization of natural soil organic matter or crop residues.

After application, fertilizer and manure N undergo several processes of biochemical and chemical transformations and transport that form the nitrogen cycle. Major transformation and transport processes in the nitrogen cycle include: hydrolysis, volatilization, immobilization in soil organic matter, sorption and fixation by soil clay minerals, mineralization (release of ammonium from soil organic matter), nitrification of ammonium to nitrate, denitrification of nitrate (leading to release of nitrogen gas, N_2, from soil), plant uptake (assimilation), leaching, runoff, and erosion. Runoff and erosion primarily transport sediment-bound organic N forms associated with soil organic matter. Leaching of nitrogen involves nitrate primarily, but not exclusively; ammonium and organic N forms tend to be bound to soil particles and are not very mobile (Jackson et al. 1973; Bottcher et al. 1981; Burwell et al. 1977). Dissolved organic matter and various soluble humic and fulvic acids containing organic forms of nitrogen also can be leached through soil, but in less significant quantities than nitrate. Throughout the MRB, leached nitrogen can be transported to surface waters via subsurface flow of ground water and tile drainage, with the latter being the more important mechanism.

3.2.4.2 NITRATE MANAGEMENT STRATEGIES

Nitrate loadings to surface waters in the Upper Mississippi River and Ohio River sub-basins occur primarily by infiltration of water beyond the crop-rooting zone and into deeper soil layers, where it is collected by subsurface tile drains. In other sub-basins (especially the Lower MRB), the primary pathway for nitrate loading to surface waters is ground-water seepage and irrigation return flow. Reductions of nitrate loading to surface waters in the Mississippi River Basin can be achieved: (1) by reducing the concentration of nitrate in drainage or irrigation return flow waters entering the river and (2) by reducing the volume of drainage or irrigation water.

Drainage losses of nitrate from nonirrigated cultivated soils most commonly are caused by a combination of heavy precipitation or snowmelt and application of N-containing fertilizers or manure in excess of crop removal requirements. Excess application of fertilizer N typically occurs when organic N sources (animal wastes or legumes) are not properly credited, when soil in-organic N sources are not properly credited, or when crop yield goals are considerably larger than potential yields (Keeney 1987). The most important strategy for reducing nitrate losses in drainage water is to apply N fertilizer and manure at rates and times consistent with crop uptake requirements (Gast et al. 1974, 1978).

When nitrate accumulates in soil during a poor growing season with low crop yields (because of lack of precipitation and/or excessive heat), the residual soil N often can be removed before it is lost by leaching with fall and winter cover crops (rye grass) or a succeeding crop (soybeans or alfalfa) that receives no additional N. Establishment of cover crops is problematic in northern latitudes due to cold climatic conditions and a short growing season.

The proper application rate for fertilizer N depends primarily on the expected uptake of N by the crop minus that supplied by the soil. Crop uptake of N can be estimated from the amount of N uptake per unit of yield, the fraction of N fertilizer uptake by the crop, and the crop yield goal (Bock and Hergert 1991), which is the anticipated long-term average crop yield for specific soil types within a field. Fertilizer uptake efficiencies for corn are 55–70%, and N requirements range from 1.43 to 1.82 pounds N per bushel of yield (Bock and Hergert 1991). Soil N supply can be estimated from a test of soil residual N, the rate of mineralization of soil organic matter, and the carryover from N_2-fixing crops grown in previous years (Meisinger and Randall 1991). In addition, the recommended rate of fertilizer is reduced by a nitrogen credit for manure that will be applied to the soil (Schepers and Mosier 1991). Land-grant university guidelines for N fertilizer recommendations have been developed for every state in the Mississippi River Basin.

The magnitude of reductions in nitrate losses to drainage or leaching that can be achieved by improved management depends upon many site- and management-specific factors. The former include the characteristics of the climate, soils, and cropping system. Losses also depend upon the specific improvements in management, which may include the following options: (1) calibrating fertilizer and manure application equipment, (2) applying rates of N fertilizer that are consistent with fertilizer rate guidelines developed by the land-grant universities, (3) switching from fall to spring or split applications, (4) switching from broadcast to banded or incorporated application methods, and (5) applying nitrification inhibitors.

Improved management also could involve using yield monitors to establish realistic goals for crop yield; adopting pre-plant or pre-sidedress soil nitrate testing and variable-rate applications of nitrogen; using stalk nitrate tests or hand-held chlorophyll meters to identify crop N deficiencies; and accounting for N credits from organic matter, crop residues, or manure applications. Agronomic management techniques can maximize crop yields and reduce nitrate losses—e.g., adopting crop rotations that include legumes; improving weed, insect, and disease management techniques; and planting early-season crops and high-yield crop cultivars. Improved water management techniques include adopting controlled drainage or sub-irrigation methods, switching from furrow irrigation to surge irrigation or sprinkler irrigation with fertigation (in which soluble fertilizer is added directly to irrigation water), and using irrigation scheduling techniques. Finally, nitrate losses from drainage and irrigation systems can be reduced at and beyond the edge of the field by intercepting and treating the drainage or irrigation water in grassed waterways, sediment basins, wetlands, controlled-flow ditches, vegetative buffers, and riparian buffer strips.

Control of depth (of the soil column) to the water table has a good potential for reducing nitrate losses through artificial tile-drainage systems. Such control can be achieved by managing the spacing and depth of tile drains and by controlling structures on the tile drain outlets or sub-irrigation. The concept here is to optimize depth in order to achieve a balance between acceptable crop yields and reduction in nitrate losses by denitrification and crop uptake. In Indiana, nitrate losses from a field in continuous corn production through a subsurface drainage system with a spacing of 20 m were 27% lower than losses through drainage systems with a spacing of 10 m and 46% lower than losses through a drainage system with a spacing of 5 m (Klaivko et al. 1991). In Ohio, nitrate losses were 21% lower from a drainage system with a depth of 0.4 m, compared with losses from a system with a depth of 0.9 m (Schwab et al. 1985). Studies in North Carolina (Evans et al. 1989) and Iowa (Zucker and Brown 1998) showed that water tables at a controlled depth of 0.6–1.0 m reduced nitrate losses through tile drains by 45–54%, compared with losses through conventional drainage systems. Studies with sub-irrigation in Michigan (Fausey et al. 1995) reported reductions in nitrate losses of 58–64%, compared with conventional drainage systems.

Nitrate losses from tile-drained fields can be reduced at the edge of the field and beyond using various strategies, including (1) denitrification of nitrogen in the drainage water with wetlands, (2) uptake of N in grass or forest buffer strips, and (3) enhanced denitrification through reductions in the flow rate of drainage water in surface ditches. Research in Iowa and Illinois (Zucker and Brown 1998) and North Carolina (Chescheir et al. 1992) showed that nitrate loads can be reduced by 60–90% by wetland treatment. For adequate nitrate removal, 1–5% of the contributing watershed area must be in wetlands, and treatment efficiency is greatest during spring and summer. Grass buffer strips 4–18 m long have been shown to reduce nitrate loads of water flowing through the strips by 54-80% (Dillaha et al. 1989; Srivastava et al. 1996). Similarly, riparian forest buffers have been shown to reduce nitrate loads in drainage water flowing through them by 90–96% (Yates and Sheridan 1983; Gilliam 1994).

3.2.4.3 ORGANIC NITROGEN MANAGEMENT STRATEGIES

Many of the practices described in the preceding section also will help to reduce discharges of organic N from agricultural areas. This is especially true regarding the use of grass and woodland buffer strips and receiving wetland areas, which help to prevent eroded soil from reaching waterways. Other practices to control organic N runoff involve the control of soil erosion from croplands and include reduced tillage, contour cropping, terracing, and winter cover crops. These practices provide the additional benefit of reducing phosphorus in runoff.

3.2.4.4 PROCESSES AFFECTING PHOSPHORUS TRANSPORT

Phosphorus is transported from agricultural lands to surface waters primarily by runoff and erosion, as well as by direct discharge from animal waste storage lagoons. Phosphorus is an essential plant nutrient. The amount required for crop production is determined by a variety of soil-extraction procedures that measure “plant-available P.” When available P levels at the soil surface exceed threshold levels at which there is no further response by the crop (Sharpley et al. 1994), the potential for P losses to surface waters increases. The critical threshold varies by state and crop, but generally ranges from 25 to 100 mg P/kg for the Bray-1 soil extraction and from 35 to 120 mg P/kg for the Mehlich-1 extraction (Sharpley et al. 1994).

The actual amount of phosphorus loss in runoff or erosion from cultivated fields depends upon several factors (S.J. Smith et al. 1993). The main processes for soluble P loss from cultivated lands involve desorption, dissolution, and extraction of phosphorus from soil and crop residues (Sharpley et al. 1994). Particulate P losses are associated primarily with soil erosion. Losses generally occur during intense spring and summer rainstorms, especially on soil vulnerable to surface runoff and having sparse crop residue or plant canopy cover. Phosphorus losses are exacerbated when areas of high surface runoff and erosion potential coincide with areas of high soil P that have resulted from repeated applications of P (as fertilizer or manure) in excess of crop needs (Sharpley et al. 1996, 1998). Significant losses can also occur prior to the growing season during spring snowmelt runoff. Losses of P in the former case are dominated by sediment-bound or organic matter-bound particulate P carried from the field by erosion. Losses in the latter case may be dominated by dissolved P contained in runoff that passes through decaying crop residue or animal manure.

The inherent soil and landscape features that control runoff and erosion (e.g., soil permeability and erodibility, slope length and steepness) play important roles in determining the actual rate of P loss. In general, the locations most vulnerable to P losses are those having high sediment delivery ratios in close proximity to surface waters (Gburek and Sharpley 1998). Losses of P by leaching through the soil and entry into subsurface tile-drainage systems generally are much smaller in magnitude than surface losses by runoff and erosion. Exceptions can occur on sandy soils with a history of excessive P fertilizer and manure applications (Sims et al. 1998).

The total loss of P from agricultural lands can be partitioned into particulate and dissolved P. The latter includes inorganic (orthophosphate), as well as organic P forms, but inorganic P predominates in most cases. Inorganic dissolved P is immediately bioavailable. The short-term bioavailability of particulate P has a very wide range of 10–90% (Daniel et al. 1998). Particulate P may become bioavailable over a long period as it travels through the aquatic system (Sharpley 1993).

3.2.4.5 PHOSPHORUS MANAGEMENT STRATEGIES

The strategies for reducing P losses to surface waters from cultivated lands differ in some respects from those used to reduce N losses. With phosphorus, the most important strategy is to control runoff and erosion. A secondary strategy that works with both N and P is to intercept and treat the discharge from cultivated fields using grassed waterways, grass buffer strips, riparian forest buffers, and wetlands. The discharge intercepted for P removal, however, should be primarily from surface runoff and erosion, whereas the intercepted discharge for nitrate removal should be primarily from subsurface tile-drainage effluent.

Phosphorus losses to surface waters can be significantly lowered by reducing runoff and erosion from cultivated fields (Sharpley et al. 1994). The primary strategy involves promoting crop-residue cover during the early portion of the growing season by using conservation tillage. Other strategies for controlling erosion, such as contour tillage, terracing, or cover crops, also can reduce P losses to surface waters. Typically, the percentage reductions achieved for soil erosion are greater than the reductions in total P losses, because erosion control reduces dissolved P losses much less than it reduces particulate P losses (Daniel et al. 1998).

During the last several decades, soil-available phosphorus levels have increased significantly in many portions of the Upper Midwest (Sharpley et al. 1994; Randall et al. 1997) because of long-term applications of P fertilizer and manure. Reducing very high levels of available P in soils to threshold agronomic levels may require several decades with no further P additions to the soils (Sharpley et al. 1994). For this reason, it is important to prevent the development of excessively high levels of soil-available P by properly managing fertilizer and manure application rates. This can be achieved by regularly testing soils for available P levels and applying fertilizer at rates consistent with widely published university guidelines. In addition, credits can be included for the P supplied in manure, and fertilizer recommendations can be reduced accordingly. Phosphorus losses in runoff and erosion also can be reduced by incorporating fertilizer or injecting manure below the soil surface to decrease P desorption, dissolution, and extraction.

These P management strategies should be targeted to critical source areas of a watershed, where high surface runoff and erosion potentials coincide with elevated soil P (Gburek and Sharpley 1998). This recommendation is based on the observation that as much as 90% of the annual P exported from watersheds can occur from less than 10% of the land area during a relatively few large storms. For example, more than 90% of the annual total P export and more than 75% of the annual water discharged from watersheds in Ohio (Edwards and Owens 1991) and Oklahoma (S.J. Smith et al. 1991) occurred during one or two severe storms. Without focusing on these critical source areas, broadly applied remedial measures are likely to be an inefficient and expensive approach to reducing P exports to surface waters.

A significant difference between strategies for N and P is that N losses can occur from any location in the watershed, while areas prone to surface runoff contribute most to P losses. Hence, remedial strategies for N can be applied to the whole watershed, whereas the most effective P strategy would be to prevent excessive buildup of P across the whole watershed and reduce surface runoff from critical areas that have a high potential for exporting P to surface waters.

In the past, separate strategies for N and P have been developed and implemented at farm or watershed scales. Because the chemistry and flow pathways of N and P differ in soil, these narrowly targeted strategies often lead to a decrease in one nutrient-loss pathway and to an increase in others. For example, basing manure application on crop N requirements to minimize nitrate losses in drainage water can increase soil P and enhance surface runoff losses of P (Sharpley et al. 1998; Sims et al. 1998). In contrast, reducing surface runoff losses of P via conservation tillage can enhance N leaching losses (Boesch et al. 1999; Sharpley and Smith 1994).

In addition to controlling and treating runoff and erosion and properly managing fertilizers and manure, changing animal feed practices may significantly reduce P losses to surface waters. The P content of animal feed can be matched to dietary intake requirements, thereby reducing P concentrations in animal manure (Daniel et al. 1998). An alternative strategy is to add the enzyme phytase to animal feed to increase animals' retention efficiency of feed phosphorus (Daniel et al. 1998). Finally, corn used for animal feed can be altered genetically to produce low levels of phytic acid phosphorus (Ertl et al. 1998), thereby reducing excretion of P.

Composting is another potential tool for manure management (DeLuca and DeLuca 1997). Composting increases the N:P ratio of manure, which allows the manure to more closely match crop N and P uptake requirements. Adding slaked lime or alum amendments during manure composting can reduce ammonia volatilization, the solubility of P, and the concentration of P in surface runoff. Although composting tends to increase the P concentration of manure, the volume is reduced, making it cheaper to transport. Currently, manure rarely is transported more than 10 miles from where it is produced, which leads to buildup of available P and N in the soil from repeated applications of manure in a single area. Continued manure applications should be restricted to areas where (1) soil nutrient levels are above threshold levels for protecting water quality, (2) runoff and erosion potentials are high, and (3) sensitive water bodies are nearby.

3.2.4.6 Summary of Agricultural Options for Reducing Nitrate Loads in Surface Waters

Nitrate loading to surface waters from cultivated land generally is a result of:

- heavy precipitation or snowmelt,
- subsurface drainage systems,
- high organic matter-content soils, and/or
- N fertilizer or manure applications in excess of agronomic recommendations.

Annual or seasonal nitrate losses to surface waters can be significantly reduced by a wide variety of improved management practices, as illustrated by the following:

- Subsurface tile-drainage spacings of 20 m can reduce nitrate losses by 27–46%, compared with spacings of 5–10 m.
- Water-table control strategies can reduce nitrate losses by 45–54%, compared with conventional subsurface drainage systems.
- Routing tile-drainage effluent through wetlands, grass buffer strips, and riparian forest buffers can remove nitrate loads by 60–90%, 54– 80%, and 90–96%, respectively.
- Changing from row-cropping to perennial-cropping systems can decrease nitrate losses by over 90%; however, this practice would severely reduce farm income levels.
- Planting a cover crop of rye grass can reduce nitrate losses by 29–94% during the fall and winter.
- Switching from conventional to ridge tillage and from fall to spring fertilizer application can cut nitrate losses by up to 25% and 27%, respectively.
- Testing for available N and applying N fertilizer and manure at agronomically reasonable rates can reduce nitrate losses from corn fields by 40–95%, compared with fields where N fertilizer and manure are overapplied.

The degree of reduction achieved depends on site-specific characteristics (climate, soils, cropping history), the specific management improvements, and the baseline (or initial) conditions to which the management improvements are being compared. For instance, a producer applying agronomically reasonable rates of N fertilizer and manure in the spring generally will not be able to reduce nitrate losses as much as a producer overapplying N in the fall.

From a practical standpoint, not all of these options are equally viable. Managing the rate and/or timing of N application probably is the most viable of the options; growing perennial crops and planting cover crops are probably the least viable. Adopting many of these management alternatives may be slow for a variety of reasons, including economic disincentives. Significant reductions in nitrate loads at the scale of watersheds and basins require widespread adoption of improved N management practices. Furthermore, a strategy is needed that promotes adopting improved N management practices in watersheds with the highest N loads.

3.2.5 Effects of Nutrient-Source Changes on Concentrations and Loads

The most direct and obvious effect of changes in land-management practices to reduce nutrient losses from the land and nutrient loadings to surface waters in the Mississippi River Basin will be a decrease in concentrations of nutrients in MRB rivers and streams. In turn, the changes in nutrient concentrations will induce other changes in trophic conditions and general water quality within the surface waters. The latter changes are described in sections 3.2.6 and 3.2.7.

This section examines the likely changes in nutrient concentrations and loads that may result from various nutrient-source control strategies. These changes are not simple to predict for a large, complicated drainage basin like the MRB. Because of nutrient retention and loss mechanisms that vary in importance as a function of spatial and temporal scales, it is particularly difficult to predict downstream loads that will result from management changes in the upper part of the drainage basin.

We start by briefly examining current nutrient conditions in the MRB. Our analysis of likely changes in nutrients is conducted on a regional basis, with more attention given to sub-basins that currently have high concentrations and are major contributors to the total load at the mouth of the Mississippi River. We also examine preliminary results from a large landscape–hydrologic model being developed for the entire MRB to consider effects of changes in farming practices on nutrient concentrations and loads in the river system. Finally, we review various lines of evidence to assess the extent of nutrient retention and loss in MRB rivers.

3.2.5.1 OVERVIEW OF CURRENT CONDITIONS FOR NUTRIENT CONCENTRATIONS

Geographic trends in nutrient concentrations were assessed for this report by selecting 24 water quality sampling stations (generally the mouth sites of major tributaries) from across the Mississippi River Basin. Data are from a 74-station data set (Table 3.1), and results are summarized in Figures 3.2–3.4 as frequency histograms.

Nitrate concentrations were generally below 2 mg N/L for most stations. Tributaries in the Upper Midwest generally had the highest concentrations, and rivers on the western side of the MRB tended to have the lowest. High nitrate concentrations were found in the Minnesota River (station 5330000), Iowa River (station 5465500), Illinois River (station 5586100), and near the mouth of the Missouri River (station 5587455). The Greater Miami River (station 3274600), which drains into the Ohio River, had high nitrate concentrations (the Miami drains a heavily agricultural region). However, nitrate concentrations are much lower at the mouth of the Ohio River (station 3612500). Dilution by low-nitrate rivers like the Cumberland River (station 3438220) and the Tennessee River (station 3609750) explains these trends.

Total nitrogen (TN) concentrations (Figure 3.3) generally were distributed similarly to nitrate, but stations in Nebraska, Kansas, Oklahoma, and Texas had a significant number of values in the 2–4 mg/L range, even though nitrate levels generally were below 2 mg N/L in these areas. Organic N is a significant nitrogen component in these rivers; ammonium concentrations (not shown) generally are very low (< 0.2 mg/L), compared with nitrate and TN values. Total phosphorus (TP) generally was below 0.2 mg/L at most stations, but some exceptions exist (Figure 3.4) in the Upper Midwest and in the same Great Plains states that had high TN values.

In summary, the trends shown in Figures 3.2–3.4 support the more detailed findings of others that the Corn Belt has the highest nutrient loadings in the MRB. A more complete analysis of this subject may be found in the Task Group 3 report (Goolsby et al. 1999).

TABLE 3.1. Locations and ID numbers for 74 selected water quality sampling stations in the Mississippi River Basin.

River System/Station Name	ID #	River System/Station Name	ID #
Ohio River System		**Missouri River System**	
Allegheny R. at New Kensington, PA	3049625	Bighorn R. at Bighorn, MT	6294700
Cumberland R. near Grand Rivers, KY	3438220	Cheyenne R. at Cherry Creek, SD	6439300
G. Miami R. at New Baltimore, OH	3274600	Grand R. near Sumner, MO	6902000
Green R. near Beech Grove, KY	3321230	James R. near Scotland, SD	6478500
Kanawha R. at Winfield, WV	3201300	Kansas R. at DeSoto, KS	6892350
Kentucky R. at Lock 2, at Lockport, KY	3290500	Milk R. at Nashua, MT	6174500
Monongahela R. at Braddock, PA	3085000	Missouri R. at Garrison Dam, ND	6338490
Muskingum R. at McConnelsville, OH	3150000	Missouri R. at Pierre, SD	6440000
Ohio R. at Cannelton Dam, KY	3303280	Missouri R. at Fort Benton, MT	6934500
Ohio R. at Greenup Dam nr. Greenup, KY	3216600	Missouri R. at Fort Randall Dam, SD	6453000
Ohio R. at Markland Dam nr. Warsaw, KY	3277200	Missouri R at Hermann, MO	6934500
Ohio R. at Benwood near Wheeling, WV	3112510	Missouri R. at Virgelle, MT	6109500
Ohio R. at Dam 53 near Grand Chain, IL	3612500	Missouri R. below Fort Peck Dam, MT	6132000
Scioto R. at Higby, OH	3234500	Missouri R. near Culbertson, MT	6185500
Tenn. R. at Hwy. 60, near Paducah, KY	3609750	Missouri R. near Landusky, MT	6115200
Tenn. R. at Pickwick Landing Dam (LL), TN	3593005	Osage R. below St. Thomas, MO	6926510
Tennessee R. at S. Pittsburg, TN	3571850	Platte R. at Louisvillem, NE	6805500
		Yellowstone R. at Forsyth, MT	6295000
Mississippi River Mainstem		Yellowstone R. near Sydney, MT	6329500
Mississippi R. at New Orleans, LA	7374508	Yellowstone R. near Miles City, MT	6296120
Mississippi R. at Belle Chasse, LA	7374525		
Mississippi R. at Clinton, IA	5420500	**Southern Plains System**	
Mississippi R. at Keokuk, IA	5474500	Arkansas R. at David Terry Land, AR	7263620
Mississippi R. at Memphis, TN	7032000	Arkansas R. at Ralston, OK	7152500
Mississippi R. at Ninninger, MN	5331570	Arkansas R. at Tulsa, OK	7164500
Mississippi R. at St. Paul, MN	5331000	Canadian R. at Calin, OK	7231500
Mississippi R. at Thebes, IL	7022000	Canadian R. near Whitefield, OK	7245000
Mississippi R. at Vicksburg, MS	7289000	Red R. at Alexandria, LA	7355500
Mississippi R. at Winona, MN	5378500	Red R. at Shreveport, LA	7344410
Mississippi R. below Grafton, IL	5587455	Red R. at Indes, AK	7337000
Mississippi R. near Royalton, MN	5267000	Red R. near Simmesport, LA	7355601
Mississippi R. near St. Francisville, LA	7373420	St. Francis Bay at Riverfront, AR	7047900
		White R. at Clarendon, AR	7077800
Upper Mississippi River System			
Chippewa R. at Durland, WI	5369500	**Lower Mississippi River System**	
Illinois R. at Marseilles, IL	5543500	Kaskaskia R. near Venedy Station, IL	5594100
Illinois R. at Valley City, IL	5586100	Atchafalaya R. at Melville, LA	7381495
Iowa R. at Wapello, IA	5465500	Big Black R. near Bovina, MS	7290000
Minnesota R. at Jordan, MN	5330000	Lower Atchafalaya R. at Morgan City, LA	7381600
Rock R. near Joslin, IL	5446500	Ouchita R. at Columbia, LA	7367640
St.. Croix R. at St.. Croix Falls, WI	5340500		

Wisconsin R. at Muscoda, WI	5407000	

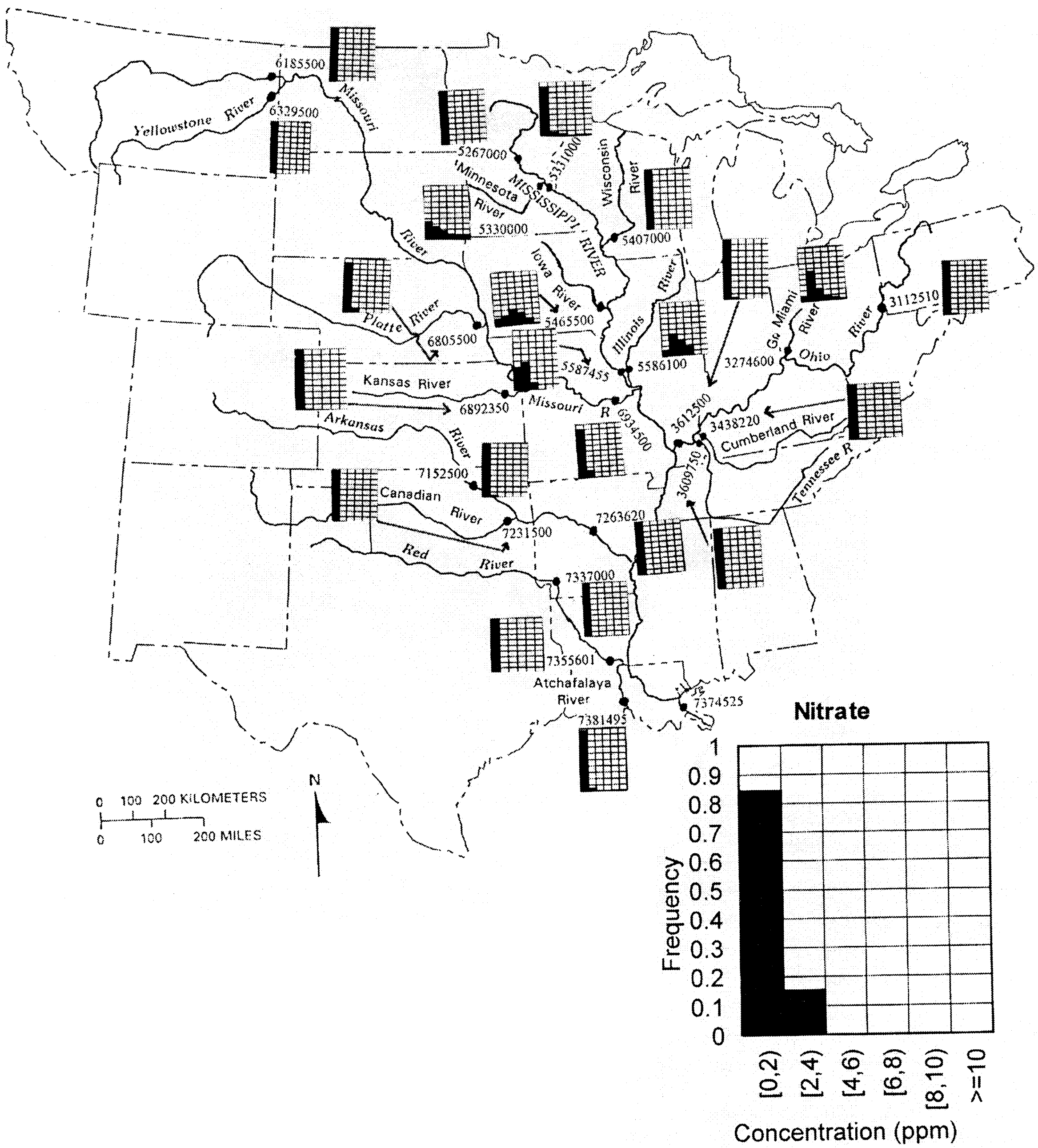


FIGURE 3.2. Frequency distribution of nitrate concentrations for selected water quality sampling stations in the Mississippi River Basin. *(Stations identified in Table 3.1; map adapted from Meade 1996; data from D. Goolsby, USGS, Denver, CO, 1998.)*

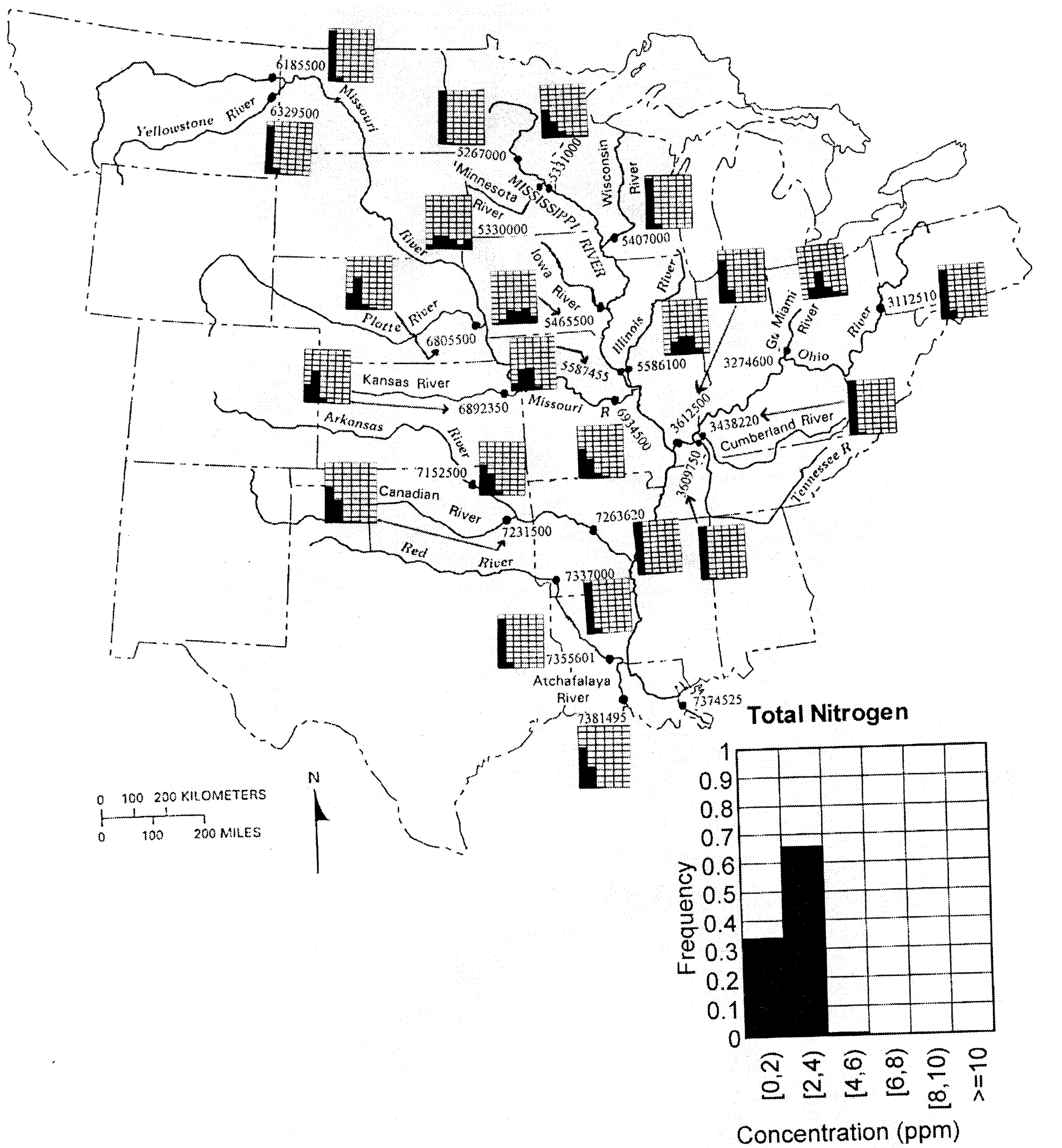


FIGURE 3.3. Frequency distribution of total nitrogen concentrations for selected water quality sampling stations in the Mississippi River Basin. *(Stations identified in Table 3.1; map adapted from Meade 1996; data from Goolsby 1998.)*

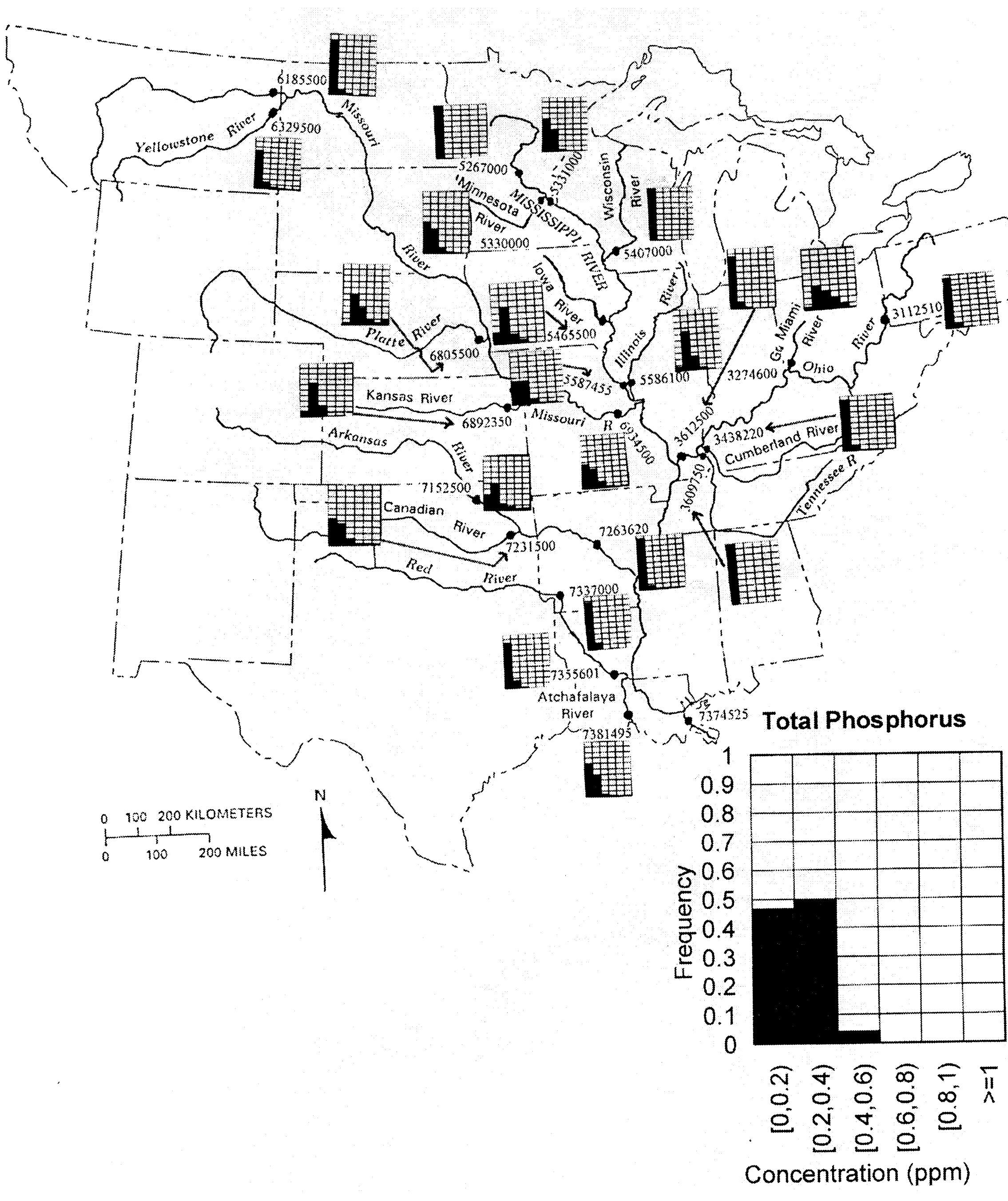


FIGURE 3.4. Frequency distribution of total phosphorus concentrations for selected water quality sampling stations in the Mississippi River Basin. *(Stations identified in Table 3.1; map adapted from Meade 1996; data from Goolsby 1998.)*

3.2.5.2 CASE STUDY: MINNESOTA RIVER BASIN

Nitrate loading of surface waters from cultivated land in the Mississippi River Basin is greatly influenced by such factors as climate; soil and landscape characteristics; cropping system; fertilizer and manure application rates; artificial drainage by subsurface and surface methods; and edge-of-field management with vegetative filter strips, riparian forest buffers, and wetlands. Phosphorus loadings are controlled by the same factors, except that surface losses by erosion and runoff are much more important than losses in drainage.

Nitrate loadings in the MRB vary considerably among sub-basins. For example, the portion of the MRB above the Missouri River confluence produces about 39% of the total nitrate load for the entire MRB, even though it represents only about 14% of the MRB land area (Goolsby et al. 1999). This portion of the basin includes much of the Corn Belt and large parts of Minnesota, Iowa, Wisconsin, Illinois, and Missouri and small parts of Indiana and South Dakota (i.e., all of the Upper Mississippi and most Middle Mississippi sub-basins delineated in Figure 3.1). This is because this region, an area of moderately high mean annual precipitation, has (1) extensive row-crop and animal agriculture, (2) high application rates of nitrogen in fertilizer and manure, and (3) large areas where the soils are high in organic matter and are artificially drained.

The Minnesota River Basin (Figure 3.5) is typical of conditions prevailing throughout much of the Upper and Middle Mississippi (including the Racoon, Cedar, Illinois, Rock, and Iowa Rivers) and Ohio (including the Scioto, Wabash, and Miami Rivers) sub-basins, where nitrate loadings to surface waters are significant. It is not representative of other sub-basins (generally farther north) in the Upper MRB that have different soils and land cover (e.g., the St. Croix, and Chippewa basins). Nitrate loadings from the Minnesota River Basin and its 12 major watersheds have been extensively studied (e.g., Mulla and Mallawatantri 1997; Mulla et al. 1997); approximately 10% of the N loading to the Upper MRB originates in the Minnesota River Basin. The following information thus illustrates the factors that contribute to high nitrate loadings of surface water from cultivated land in the Corn Belt and Upper Midwest. Mulla (1998) published a similar case study on the Minnesota River Basin that focused on P losses.

The Minnesota River Basin covers an area of ~4.0 million ha in southern and central Minnesota. The soils of the basin are high in organic matter (Figure 3.6). About 40% of the soils have poor internal drainage, which resulted in extensive natural wetlands, and many of these soils have been drained by surface ditches or subsurface tile systems. As a result of widespread drainage, approximately 92% of the Minnesota River Basin is now cultivated, primarily with row crops using a corn and soybean rotation.

Effects of precipitation on nitrate loadings

Mean annual precipitation in the Minnesota River Basin varies from 560 mm on the western side to over 790 mm on the eastern side. In conjunction with this is an even larger increase in the proportion of precipitation that flows into surface waters. On the western side of the basin, about 100 mm of the mean annual precipitation flow into rivers; on the eastern side, about 200 mm. The additional flow on the eastern side mainly represents water that percolates through the soil, is intercepted by subsurface and surface drainage systems, and then enters streams and rivers. As this water flows through the soil, it leaches nitrate.

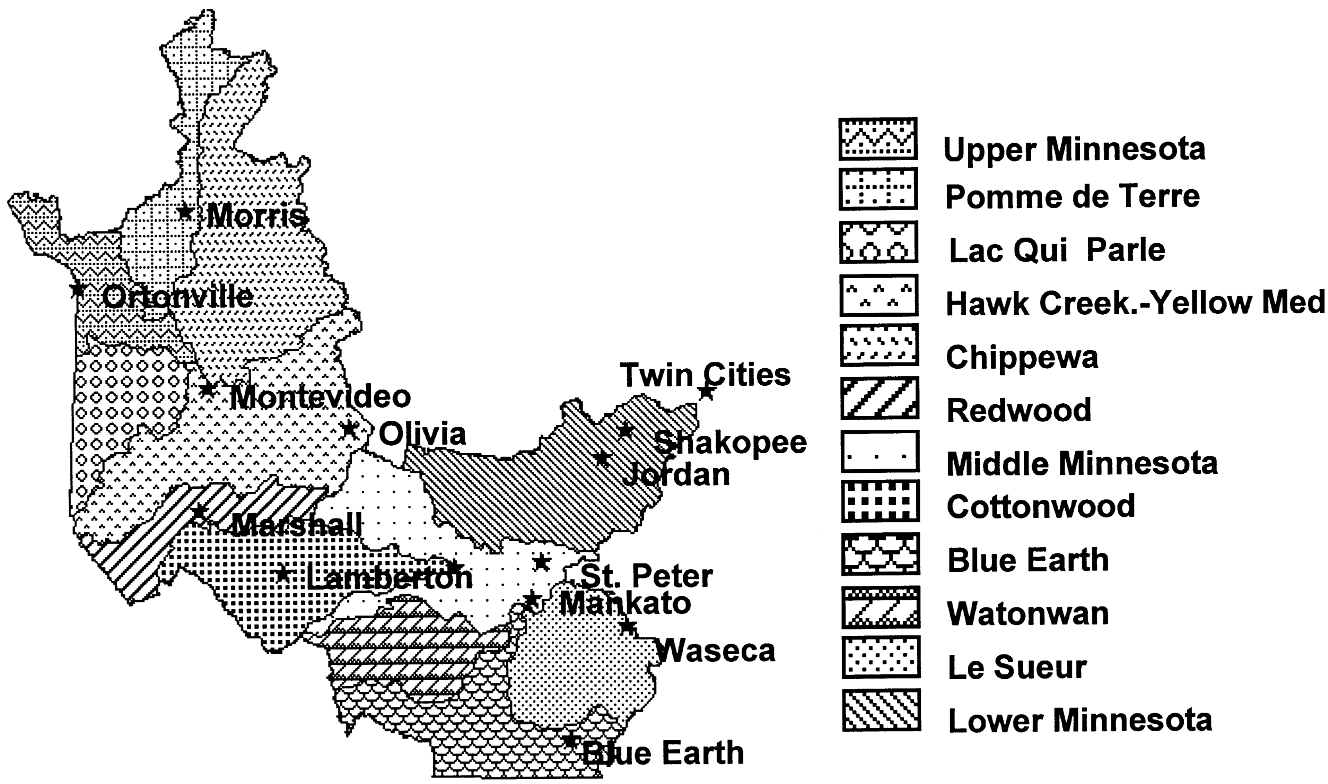


FIGURE 3.5. Map of the Minnesota River Basin (portion in Minnesota only) showing the 12 major watersheds comprising the basin.

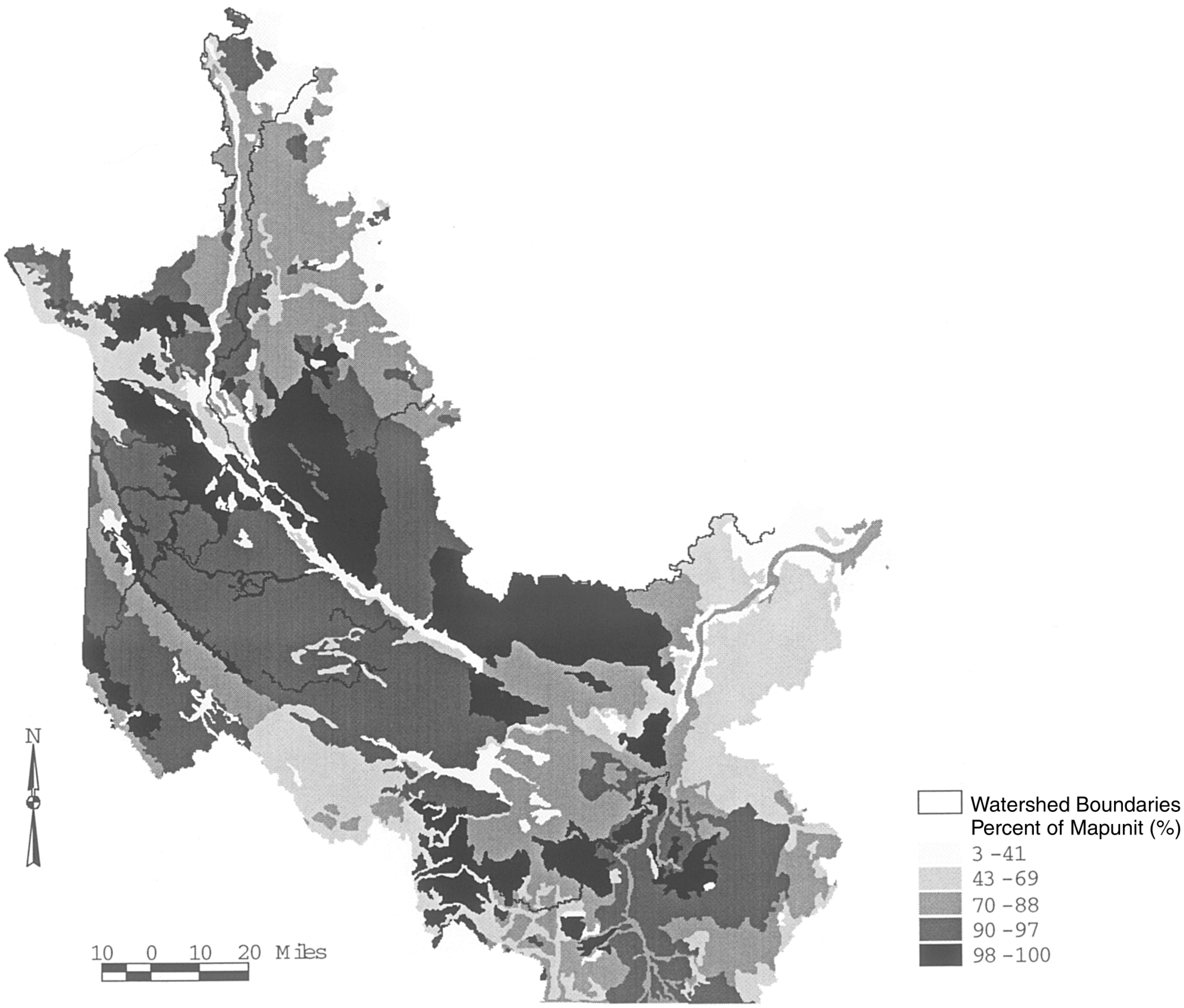


FIGURE 3.6. Distribution of surface soils in the Minnesota River Basin that contain more than 4% organic matter. NOTE: Map units represent a combination of major watersheds and agro-ecoregions. *(D. Mulla and J. Bell, unpublished, University of Minnesota, St. Paul, 1998.)*

Primarily in response to the gradient in precipitation, nitrate loadings from the major watersheds on the eastern side of the Minnesota River Basin (which has a wetter climate than the western side) are much greater than those from the major watersheds on the western side (Figure 3.7). In fact, two-thirds of the total nitrate loading from the basin originates in three of the eastern-most major watersheds (the Blue Earth, Le Sueur, and Lower Minnesota), which comprise only 31% of the basin's total area. The six western-most major watersheds, which occupy a drier climatic region, generate only 7% of the nitrate loads in the basin.

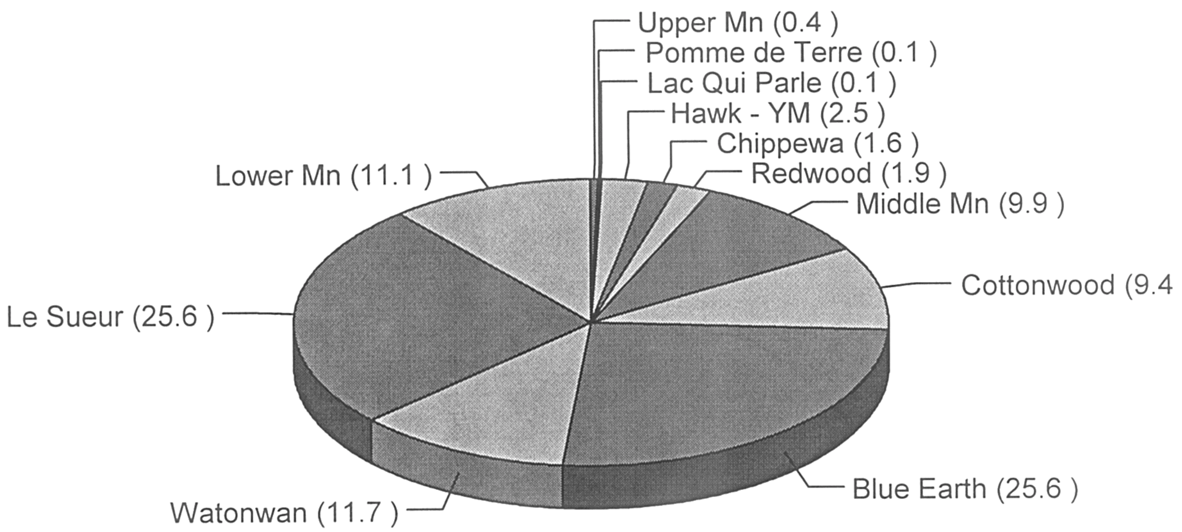


FIGURE 3.7. Percentage contributions of major watersheds in the Minnesota River Basin to the total nitrate load contributed by the river to the Mississippi. NOTE**:** See Figure 3.5 for the location of the major watersheds. *(From Mulla and Mallawatantri 1997.)*

Regional differences in nitrate yields (load per unit area) also are closely related to the gradient in precipitation (Figure 3.8). Median yields for 1977–94 vary from 0.5 to over 6 kg N km^{-2} da^{-1} (1.8 to > 21.9 kg ha^{-1} yr^{-1}) among major watersheds of the Minnesota River Basin. The mean annual nitrate yield in the basin is 3.1 kg N km^{-2} da^{-1} (11.2 kg N ha^{-1} yr^{-1}). For comparison, 1973–93 median nitrate yields for the Iowa, Illinois, Wabash, Ohio, Platte, Missouri, and Mississippi River at Royalton are, respectively, 5.5, 4.8, 3.5, 1.5, 0.1, 0.01, and 0.09 kg N km^{-2} da^{-1}. Of these, the Iowa, Illinois, and Wabash Rivers have climatic, soil, and land-use conditions that are most similar to those in the Minnesota River Basin. Although the Mississippi River at Royalton is geographically close to the Minnesota River, it generates much lower nitrate yields in surface water because it drains an area with coarse-textured, well-drained soils (nitrate leaches to ground water) and is heavily forested (with low applications of fertilizer or manure).

Precipitation in the Minnesota River Basin varies widely on both annual and monthly time scales. In recent decades, severe drought occurred during 1977 and 1988, and severe flooding occurred in 1993 and 1997. During drought years, both precipitation and river flow are abnormally low (Figure 3.9), leading to relatively small nitrate loadings. During flood years, both precipitation and river flow are abnormally high, leading to relatively large nitrate loadings. Nitrate loading of rivers can also be quite high during moderately wet years that follow dry years (see 1990–91 in Figure 3.9), as a result of the residual and mineralizable N flushed out of the soil by the increased precipitation. Thus, the greatest nitrate loadings to rivers occur in flood years, and during

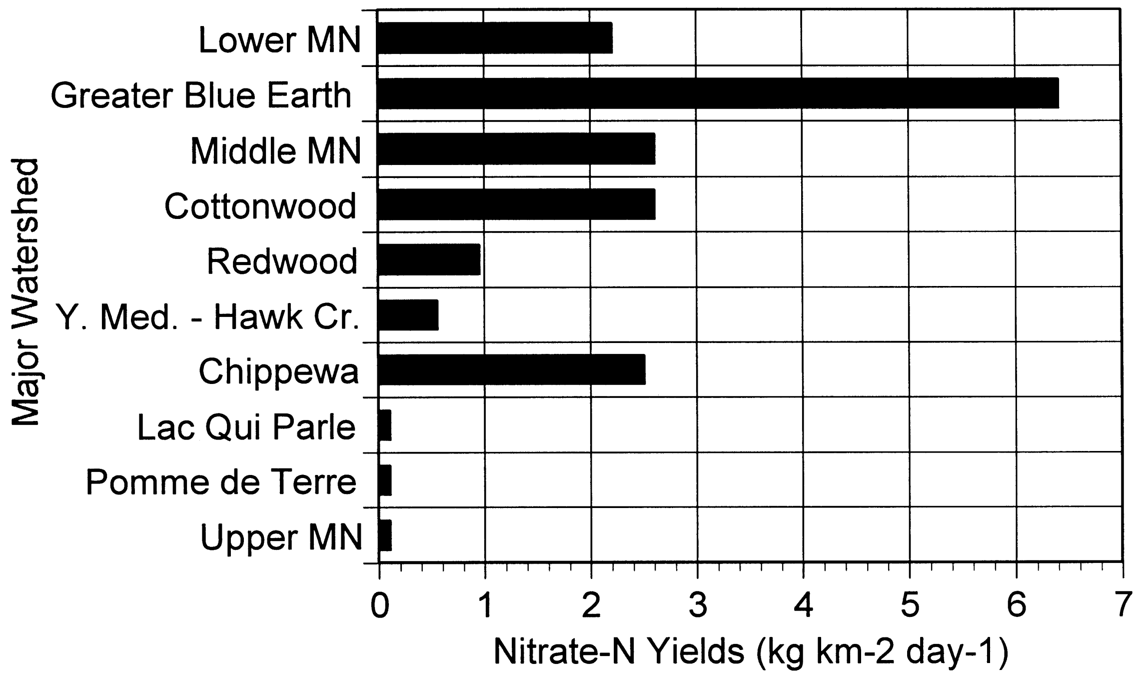


FIGURE 3.8. Nitrate yields (kg km^{-2} da^{-1}) for major watersheds in the Minnesota River Basin. NOTE: The basins are listed from top to bottom in general east to west order of their location in the basin. *(From D. Mulla, unpublished, University of Minnesota, 1998.)*

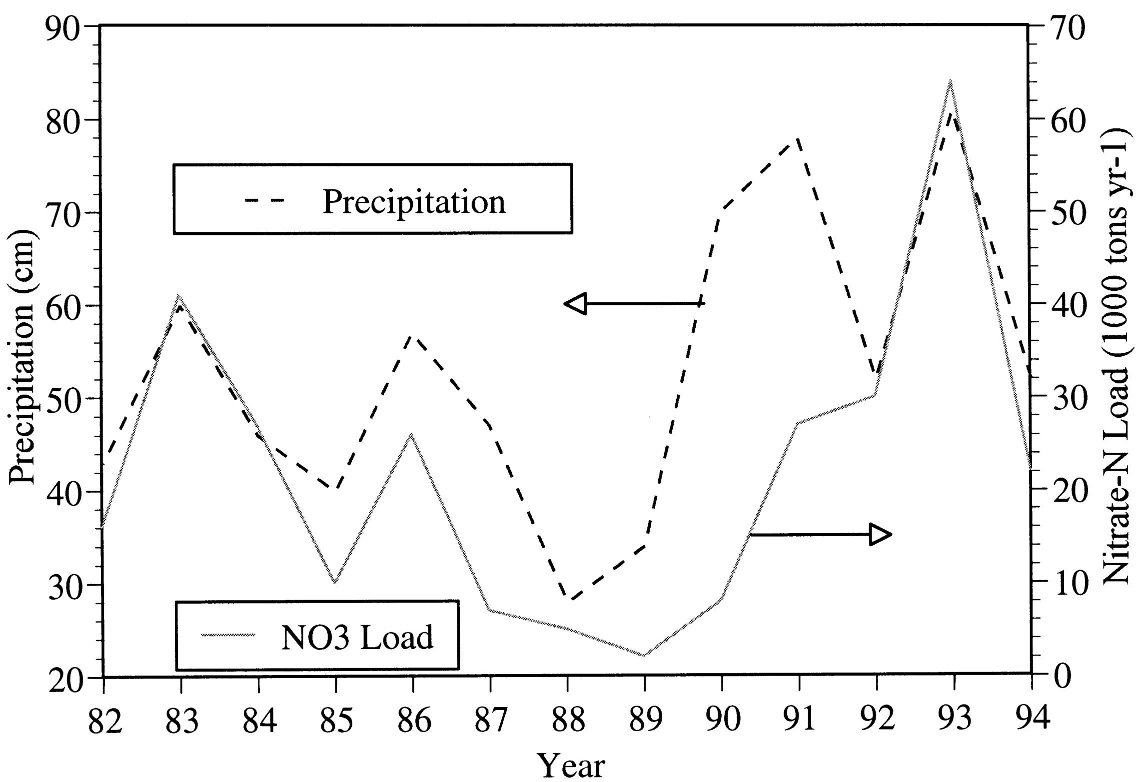


FIGURE 3.9. Annual nitrate loads of the Minnesota River generally follow annual precipitation in the basin. *(Data from D. Mulla, unpublished, 1998.)*

wet years following drought, especially where fertilizer and manure are applied to poorly drained soils with high organic matter. Most of the nitrate loading in the river generally occurs from March to July (Figure 3.10), as a result of transport of nitrogen by either snowmelt or intense rainstorms. During the relatively dry summer month of August, river flows decrease considerably, as do nitrate loadings. During other months, nitrate loadings are relatively low.

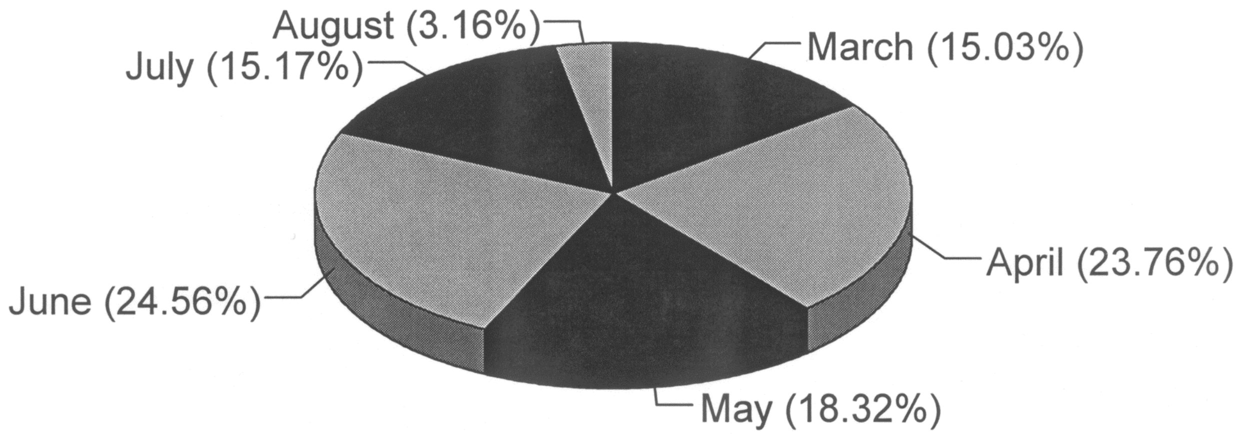


FIGURE 3.10. Distribution of monthly nitrate loadings from Minnesota's Greater Blue Earth River for March–August as a monthly percentage of the total load over the six-month period. *(From Mulla, unpublished, 1998.)*

Effects of nitrogen fertilizer and manure applications

The Minnesota River Basin illustrates the important relationships that exist between climatic variations and nitrate loadings in rivers. Land-use characteristics are of secondary importance to climatic factors. In 1996, more than 268,000 metric tons of fertilizer N were applied to cultivated land in the basin, with roughly half applied in fall. In 1992, approximately 2.3 million hogs and 0.7 million cattle were raised in the basin, and they produced manure with a nitrogen content of roughly 84,000 metric tons. Neither the pattern in fertilizer application (Figure 3.11) nor the pattern in manure application (Figure 3.12) explains as much of the variation in nitrate loadings among major watersheds of the basin as does the geographic pattern of precipitation. Judging solely by the geographical concentration of both manure and fertilizer applications in Brown, Renville, Nicollet, and Watonwan Counties, the Middle Minnesota, Cottonwood, and Watonwan watersheds should have greater nitrate loadings than other watersheds in the Minnesota River Basin. In reality, the watersheds with the greatest nitrate loadings (Lower Minnesota, Blue Earth, and Le Sueur) are not those with the greatest applied amounts of manure and fertilizer N, but are watersheds with the greatest mean annual precipitation and moderate application rates of fertilizer and manure.

Management strategies for reducing nitrate loadings in the Minnesota River Basin

The important influence of climate on nitrate loadings in tributaries and mainstem rivers in the Minnesota River Basin should not be construed to mean that land management has no influence on nitrate loadings in rivers of the basin (or indeed the entire Upper Midwest). In fact, of the controllable factors that influence nitrate loadings, land management is very important. Nutrient control in agricultural regions basically involves controlling nonpoint sources. Nitrogen, because of its many sources, transformations, and pathways, offers several opportunities for control, on either the input or the output side. Phosphorus, on the other hand, is more tied to the physical processes of runoff and erosion. Controlling runoff may minimize sediment transport, but residue cover may increase soluble P.

Possible land-management options for reducing nitrate loadings include fertilizer and manure management strategies, cropping system management, and water table management with artificial drainage.

Management practices at the edge of the field that have further potential for reducing nitrate loadings in surface waters include wetlands, vegetative buffer strips, forested riparian buffer strips, and flow-control structures in drainage ditches. Some examples of the effects of these practices on nitrate losses to surface waters in the Minnesota River Basin follow.

Mineralization of soil organic matter. Mineralization of nitrogen from soils with high concentrations of organic matter can contribute significant quantities of nitrate to outflows from artificially drained fields. As a rule of thumb, each percent of organic matter content mineralizes 34 kg ha^{-1} yr^{-1} of nitrate-N, which is available for plant uptake or leaching.

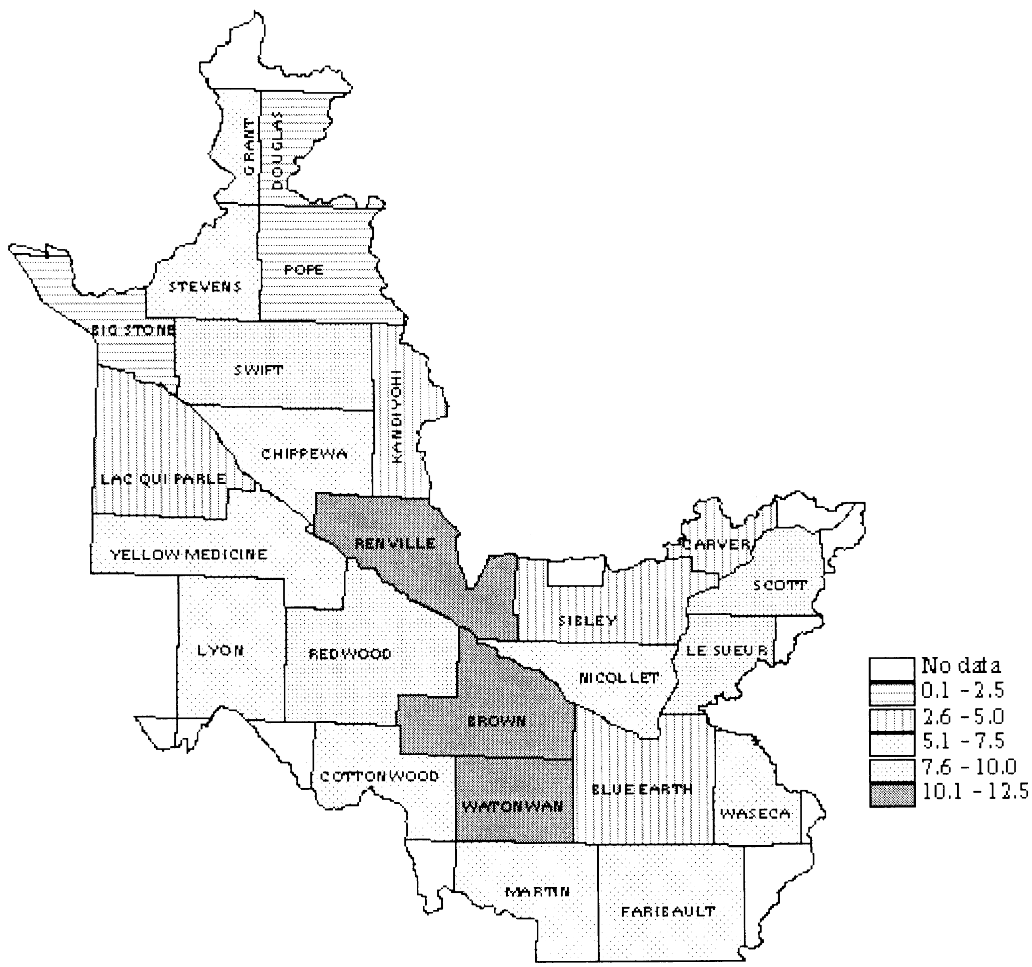


FIGURE 3.11. Fertilizer use (metric tons/km) by county in 1996 in the Minnesota River Basin.

In a five-year study in the Cottonwood watershed, Gast et al. (1978) applied only 20 kg N/ha to continuous corn plots that had not received fertilizer or manure for 10 years prior to the study. Annual flow-weighted nitrate concentrations in tile-drainage effluent were high, ranging from 13 to 28 mg N/L in years without drought. This shows that natural sources of nitrate in tile drainage can be significant for soils with high organic content. Research in Iowa (Hatfield 1995) supports this conclusion.

In addition, recently reported regional mass-balance studies on N (Burkart and James 1998) indicate that N mineralized from soil organic matter (7 million tons) is a larger potential source than inorganic fertilizer (6.5 million tons) or manure (3.5 million tons) for the MRB as a whole. Inorganic fertilizer and manure represent only about 40% of the agricultural nitrate sources (soil organic matter, atmospheric deposition, and point sources represent the other 60%), adjusted for storage and application losses, in the entire basin. N removal by crop production was estimated at 7.2 million tons. This mass-balance information should not be construed as an indication that crops remove all of the inorganic N applied, because uptake efficiencies of inorganic N rarely exceed 60%. Much of the N removed by the crop originates in organic sources, including manure and soil organic matter. N in soil organic matter, in turn, originates partly from the N applied as inorganic fertilizer or organic manure. The main point from these estimates is that a significant proportion of the N loading to surface waters is from mineralization of soil organic matter.

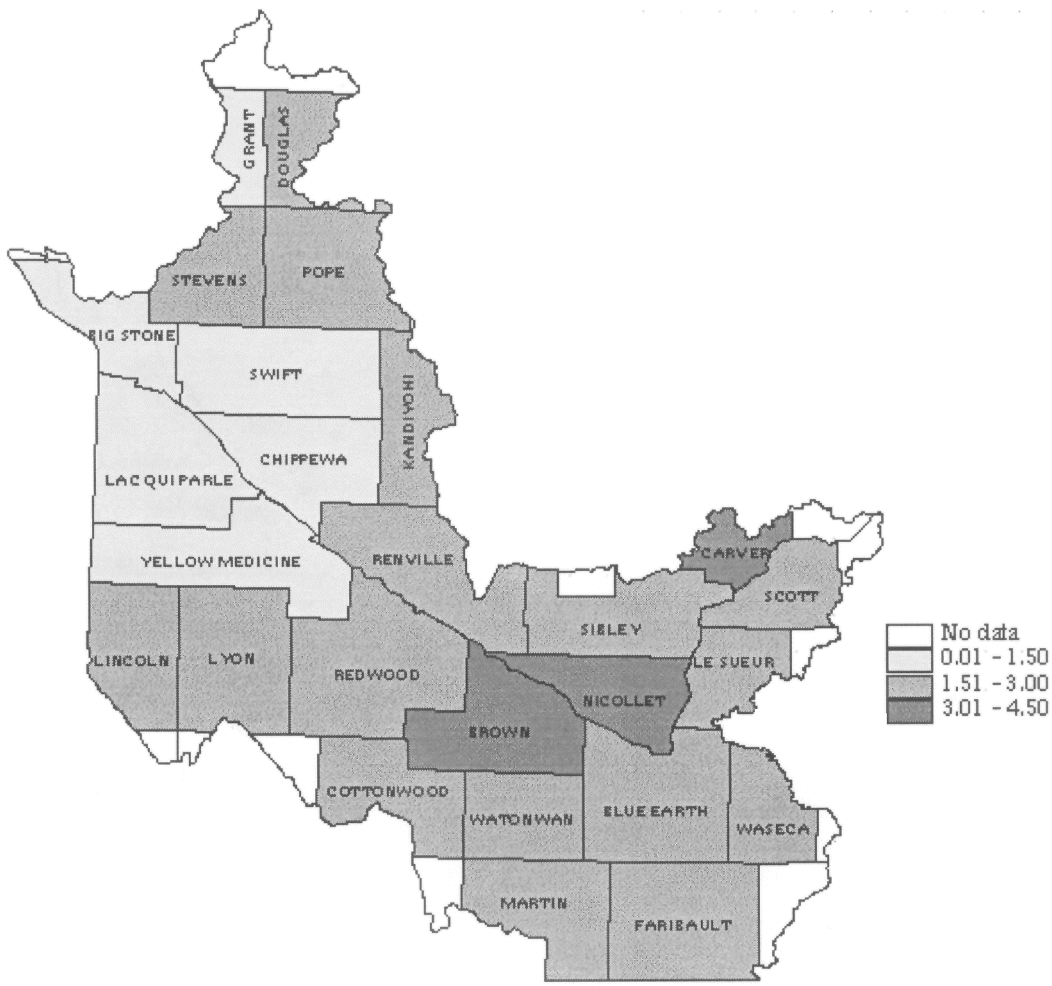


FIGURE 3.12. Animal manure use (metric tons/km) by county in the Minnesota River Basin.

Cropping systems. Cropping systems can have a significant impact on nitrate losses through tile-drainage systems (Baker and Melvin 1994; Logan et al. 1980; Weed and Kanwar 1996). Based on a four-year study in the Cottonwood major watershed, Randall et al. (1997) found that nitrate losses in tile drainage were reduced by 96–98% in perennial cropping systems, compared with the losses from row-crop systems. Nitrate losses under continuous corn, corn in a corn–soybean rotation, or soybean in a soybean–corn rotation ranged from 202 to 217 kg N/ha, whereas nitrate losses under continuous alfalfa or continuous grass ranged from 4 to 7 kg N/ha.

Compared with row-crop systems, perennial-cropping systems have such low nitrate losses because they require significantly lower amounts of N fertilizer and have greater water and nutrient uptake. Although perennial-cropping systems have a beneficial impact on the environment, they generally are not as profitable for farmers as row-cropping systems. In other areas of the Mississippi River Basin, a cover crop of rye grass was shown to reduce nitrate losses to surface waters by 29–94% from November to May (Sainju and Singh 1997). This method of controlling nitrate losses is promising for the noncrop period.

Conservation tillage. Conservation tillage is widely practiced in the Minnesota River Basin and the Upper Midwest, primarily to control soil erosion and reduce farming costs. Erosion is reduced because the crop residue reduces runoff and increases infiltration. Some scientists have been concerned that the increased infiltration could lead to increased leaching losses of nitrate.

The effects of the tillage method on nitrate losses by leaching and drainage were studied on poorly drained, Webster clay loam soil by Randall and Iragavarapu (1995) in the Le Sueur major watershed of the Minnesota River Basin. They found no significant differences in nitrate losses in drainage water between moldboard-plowed and no-till plots, and significant improvements in crop yield with conventional tillage versus no tillage.

In Iowa, Weed and Kanwar (1996) studied nitrate losses during three years of subsurface drainage on moldboard-plowed, chisel-plowed, ridge-till, and no-till plots in corn under a corn–soybean rotation. They found that nitrate losses in the ridge-till and no-till plots were reduced by about 25% compared with losses from the moldboard- and chisel-plowed plots.

Rate and timing of fertilizer application. Adjusting the rate and timing of N fertilizer application generally has more potential for reducing nitrate leaching than any other agronomic management decision. In a six-year study with continuous corn at Waseca in the Le Sueur major watershed, Buzicky et al. (1983) found that tile-drainage losses of nitrate were reduced by 27% when ammonium sulfate was applied in the spring instead of the fall. Nitrate losses were reduced by 25% when the application rate was 134 kg N/ha rather than 202 kg N/ha. Nitrate losses thus can be reduced substantially by applying lower rates of N fertilizer in the spring, in contrast to greater rates in fall.

In the same study, a control plot receiving no N fertilizer lost 62–79% less nitrate through drainage than the plots receiving fall or spring applications of 134 or 202 kg N/ha. Corn yields in the control plots, however, were reduced by 50 and 61%, respectively, compared with yields in plots receiving 134 or 202 kg N/ha. Continued profitability of corn production systems is heavily dependent on N fertilizer applications. Reductions in N fertilizer applications below agronomically recommended rates will reduce farm profits seriously. The economically optimum N rate depends on many factors, including soil type and precipitation (Oberle and Keeney, 1990), and on such external factors as the prices of fertilizer and corn (Bock and Hergert, 1991).

Randall and Iragavarapu (1995) monitored N losses for 11 years in effluent from tile drains with a 27-m spacing at Waseca on continuous corn plots with moldboard plowing. The plots consistently received 200 kg N/ha in a spring application of ammonium nitrate. Results from the plots were used to calibrate and validate the ADAPT nitrogen transport and fate model (Davis 1998), which accounts for N uptake by the crop; N transformation by volatilization, mineralization, and denitrification; and N losses in subsurface tile drainage (Alexander 1988). The model was used to investigate N losses in tile drainage with spring N fertilizer application rates of 100, 125, 150, 175, 200, and 225 kg N/ha using a 90-year climatic record from the Le Sueur watershed.

The results of this modeling effort showed that nitrate losses in drainage water were linearly related to growing season precipitation at every rate of fertilizer applied (Figure 3.13). During dry years modeled nitrate losses were small, and only small differences in nitrate losses were noted between the fertilizer rates applied. However, very large nitrate losses were found in wet years following dry years when excessive N fertilizer rates were applied. Model output indicated that during wet years nearly half of the N applied at the highest rate was lost to drainage, whereas less than 10% of the N applied at the lowest rate was lost to drainage. During wet years, a reduction in N application rate from 225 to 125 kg/ha resulted in a 92% reduction in nitrate losses in drainage. These are consistent results from Iowa (Baker and Johnson 1981), where a 44% reduction in nitrate losses occurred in four years of drainage effluent from plots receiving 95 kg N/ha, compared with the losses from plots receiving 245 kg N/ha.

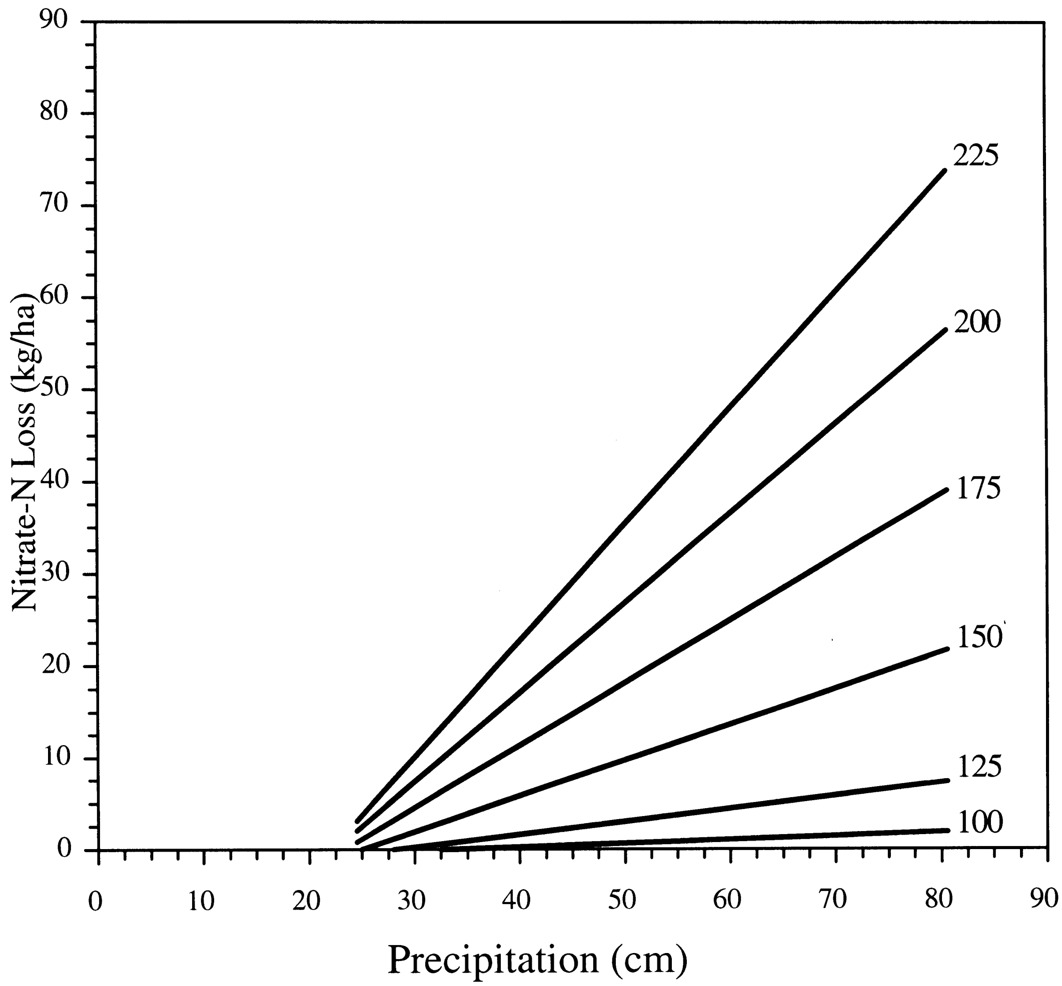


FIGURE 3.13. Model-predicted relationship between nitrate loss by tile drainage and annual precipitation at six levels of fertilizer N application. (*From Davis 1998.*)

Edge-of-field interception and treatment of drainage effluent. The validated ADAPT model described above also was used to investigate losses in nitrate beyond the edge of the field. Water quality monitoring data for 1991 and 1992 at the minor watershed scale (2,400 ha) and the major (Le Sueur) watershed scale (285,400 ha) were compared with field-scale nitrate losses estimated by ADAPT (Mulla and Addiscott 1998). Nitrate yield (kg ha^{-1}) (normalized for differences in annual precipitation and amount of artificially drained soil at each scale) showed a 41% reduction at the minor and major watershed scales, compared with losses at the field scale (Figure 3.14). This likely is due to interception and treatment (uptake and denitrification) of nitrate at the edge of the field and beyond in grassed buffer strips, forest buffer strips, wetlands, drainage ditches, and streams.

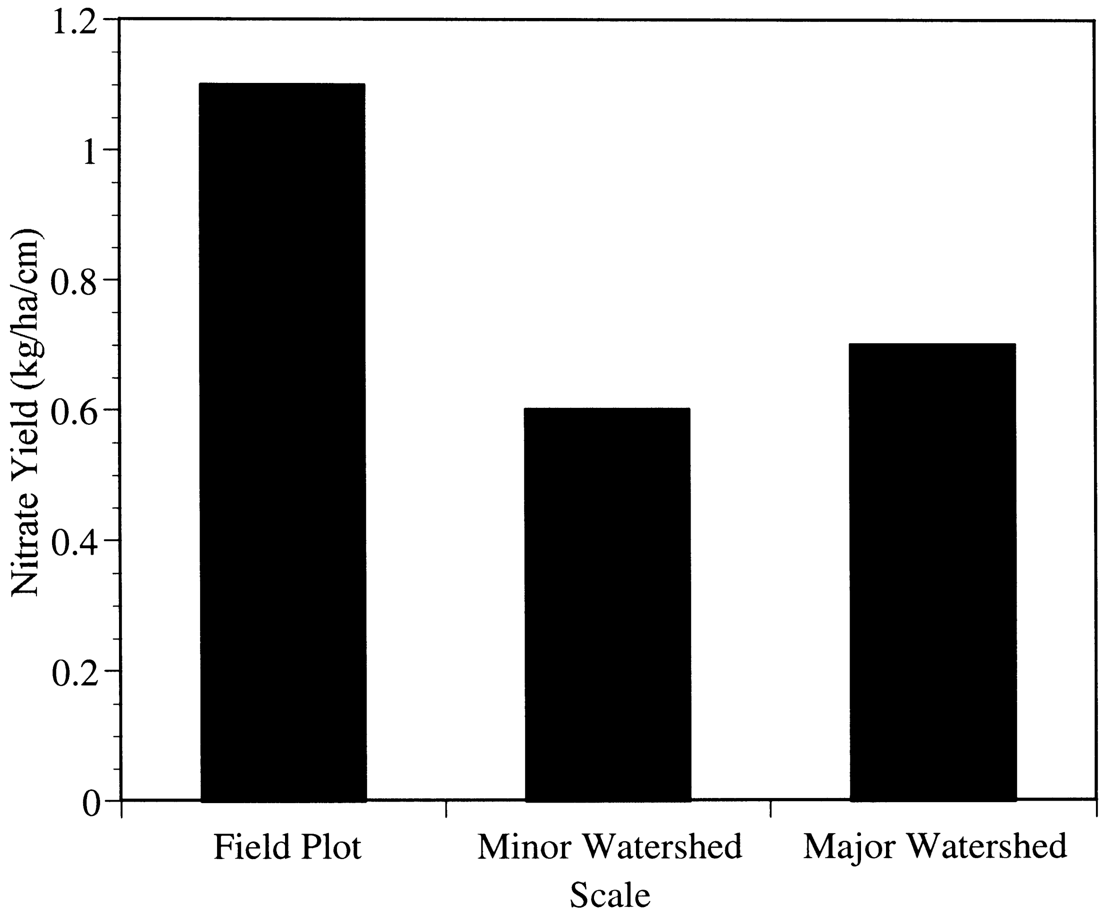


FIGURE 3.14. Nitrate yield (mass per unit area) normalized to rainfall varies with the spatial scale of the land unit. (*From D. Mulla, unpublished, 1998.*)

3.2.5.3 OPPORTUNITIES, NEEDS, AND EFFECTIVENESS OF NUTRIENT-SOURCE CONTROLS IN MAJOR SUB-BASINS IN THE MRB

To evaluate the need for improvements in agricultural practices to reduce nutrient losses from sub-basins and watersheds of the MRB, we relied primarily on an analysis of nitrate yield information (Table 3.2), i.e. mass of nitrate lost per unit area of land per time. Values in the following discussion all are reported as kg km^{-2} da^{-1}.

TABLE 3.2. Average nitrate concentrations and yields for selected watersheds in the Mississippi River Basin.[1]

Station ID	Nitrate-N *Conc.* (mg/L)	Nitrate-N *Yield* (kg km^{-2} da^{-1})	Station ID	Nitrate-N *Conc.* (mg/L)	Nitrate-N *Yield* (kg km^{-2} da^{-1})	Station ID	Nitrate-N *Conc.* (mg/L)	Nitrate-N *Yield* (kg km^{-2} da^{-1})
Ohio River System			**Missouri River System**			**Mississippi River Mainstem**		
3049625	0.65	1.28	6294700	0.31	0.04	7374508	na	na
3438220	0.45	1.24	6439300	na	na	7374525	1.43	0.75
3274600	3.94	3.40	6902000	0.95	0.93	5420500	1.86	1.73
3321230	na	na	6478500	0.56	0.04	5474500	2.53	1.85
3201300	0.58	0.90	6892350	0.86	0.11	7032000	1.57	0.92
3290500	na	na	6174500	0.25	0.01	5331570	2.62	1.18
3085000	1.07	1.60	6338490	0.15	0.01	5331000	na	na
3150000	na	na	6440000	na	na	7022000	2.48	0.90
3303280	1.35	1.81	6090800	0.23	0.09	7289000	1.48	0.91
3216600	0.95	1.56	6453000	na	na	5378500	1.53	1.20
3277200	1.35	1.54	6934500	1.25	0.23	5587455	2.66	2.40
3112510	0.92	1.32	6109500	0.21	0.07	5267000	0.23	0.09
3612500	0.98	1.46	6132000	na	na	7373420	1.47	0.71
3234500	3.47	3.79	6185500	na	na			
3609750	0.28	0.56	6115200	0.27	0.06	**Southern Plains System**		
3593005	0.36	0.60	6926510	0.40	0.40	7263620	0.35	0.16
3571850	0.34	0.55	6805500	1.19	0.14	7152500	0.65	0.10
			6295000	na	na	7164500	0.51	0.06
Upper Mississippi River System			6329500	0.30	0.05	7231500	0.84	0.05
5369500	0.55	0.54	6296120	0.22	0.07	7245000	0.27	0.05
5543500	na	na				7355500	0.19	0.14
5586100	4.10	4.83	**Lower Mississippi River System**			7344410	na	na
5465500	5.29	5.49	5594100	na	na	7337000	0.25	0.11
5330000	5.30	3.09	7381495	1.03	2.68	7355601	0.26	0.25
5446500	3.49	2.93	7290000	na	na	7047900	na	na
5340500	0.24	0.30	7381600	na	na	7077800	0.53	0.70
5407000	0.57	0.60	7367640	0.24	0.17			

Additional Stations			
5485000	6.54	6.49	Racoon River at Des Moines, IA
5463050	4.54	5.41	Cedar River at Cedar Falls, IA
5490600	4.31	4.49	Des Moines River at St. Francisville, MO
5474000	4.30	4.46	Skunk River at Augusta, IA
3378500	2.66	3.51	Wabash River at New Harmony, IN
7369500	0.54	0.72	Tensas River at Tendal, LA
7288800	0.40	0.40	Yazoo River at Redwood City, MS
6610000	1.11	0.17	Missouri River at Omaha, NE

[1]*Station names not shown are listed in Table 3.1.*
Source: D. Goolsby, USGS, Denver, CO, personal communication, 1998.

Extrapolation of the Minnesota River analysis to the Upper Mississippi River sub-basin

There is considerable evidence to indicate that the till-derived soils throughout the Upper MRB generally are similar with regard to hydrology and potential for delivering nitrate to surface waters. The soils of Illinois and Iowa and the Ohio River Basin are largely tiled and imperfectly drained. They are predominantly in row-crop agriculture and have similar rainfall and weather patterns. Consequently, it is reasonable to infer that the improved management practices described in the case study on the Minnesota River Basin should have generally similar effects on nutrient losses from agricultural watersheds throughout the Upper MRB. Land-use patterns and soil and hydrogeology information reported by Knox and Moody (1991) have been used in much of the following discussion.

Overall, highest nitrate yields in the MRB tend to be found in the Upper Mississippi sub-basin, and within this region highest yields occur in the Corn Belt. The highest values listed in Table 3.2 are associated with two agricultural watersheds in Iowa: the Racoon River (central Iowa) and the Iowa River (eastern Iowa). Land use in both watersheds is dominated by row-crop agriculture (corn and soybeans). The soils are high in organic N, and the systems are tile-drained. The Raccoon River is a sub-basin of the Des Moines River (DMR), which has its headwaters in southwestern Minnesota. Keeney and DeLuca (1993) reported that in 1990 the DMR watershed was about 78% cropland and that virtually all of the cropland was tile-drained. The nitrate yield of the DMR is 4.49 kg km^{-2} da^{-1} at St. Francisville (northern Missouri), which is farther down the watershed from the Racoon River. The Iowa River watershed is similar in land use: a somewhat older till soil, but still intensively farmed and high in organic N. The Iowa River yield is 5.48 kg km^{-2} da^{-1}. Only the Greater Blue Earth watershed of the Minnesota River Basin has a nitrate yield similar to that of the Iowa rivers (~6.5 kg km^{-2} da^{-1}; extrapolated from data in Randall and Mulla (1998)). The Illinois River system in the glaciated region also is similar to the Iowa rivers. The Illinois River is the fourth-highest sub-basin in the MRB in terms of nitrate yield (4.49 kg km^{-2} da^{-1}).

In contrast, forested watersheds in the Upper Midwest have low nitrate yields compared with watersheds largely in row-crop agriculture. For example, the nitrate yield for the first monitoring station on the Upper Mississippi River (at Royalton, MN, about 80 miles upstream from Minneapolis–St. Paul) yields only 0.09 kg km^{-2} da^{-1}; the watershed above this station contains some row-crop agriculture but is mostly forested. The Wisconsin River at Muscoda, and the St. Croix River at St. Croix Falls, WI, also are quite low. Both stations drain watersheds with a high proportion of forested area. The low nitrate yields from watersheds with substantial forest coverage suggest that such areas are of lower priority (compared with watersheds dominated by row-crop agriculture) relative to changing management practices for nutrient-source reduction.

Nitrate yields and management needs for western sub-basins

Nitrate yields are markedly lower west of the Missouri. The Missouri River at Hermann, MO, has a nitrate yield of only 0.22 kg km^{-2} da^{-1}. The Platte River yield is only 0.14 kg km^{-2} da^{-1}, and the Kansas River yield is only 0.11 kg km^{-2} da^{-1}. These values are only about 2–3% of the yields from the Iowa, Minnesota, and Illinois basins. Nitrate yields from watersheds in the far western part of the MRB (in North and South Dakota, Montana, and Oklahoma) are all < 0.1 kg km^{-2} da^{-1}. There are at least three reasons for the low yields in western parts of the basin. First, annual rainfall is much lower than in sub-basins to the east. Second, much of the drainage in the west is deep, and the nitrate, rather than running off or going through tile drains, probably is stored in the profiles. Third, land use for agriculture is far less intensive (i.e., the percentage of land in row crops is much lower) in the Great Plains than in the Corn Belt.

Relative to hypoxia problems in the Gulf of Mexico, it would seem that these regions do not need to receive concentrated attention, although local problems may exist that may need attention. In that regard, it should be noted that all the basins and watersheds discussed above are quite large, and there are many sources of variability in the landscape. A low-yielding large basin may have numerous sub-basins where nutrient control may be warranted to protect local water bodies. Furthermore, collective control of these smaller local sources may have a significant effect on N and P yields to the Mississippi.

Eastern and southern sub-basins

Less extensive use of land for row crops and perhaps higher propensity for denitrification may explain why most of these watersheds have lower nitrate yields than those in the Upper Midwest. Yields in the Ohio River Basin (1.2–1.8 kg km^{-2} da^{-1}) are higher than those of the Tennessee River Basin and the White River (Arkansas), which have yields in the range of 0.5–0.6 kg km^{-2} da^{-1}. Agriculture is more intensive in Ohio than in Tennessee and Arkansas, and much of Ohio is also tile-drained.

While nitrate yields are relatively low in the lower basins, there are some areas of intensive (row-crop) agriculture, particularly adjacent to the Mississippi River in Arkansas, Mississippi, and Louisiana. These areas also are irrigated. Because the inputs from these areas flow directly into the Mississippi, some of these watersheds should be targeted for close examination of the potential for improved nutrient management.

3.2.5.4 LARGE-SCALE MODELING OF LANDSCAPE NUTRIENT RETENTION

Experiments and experiences conducted at the field scale and in small watersheds should not be linearly extrapolated to estimate changes in nutrient deliveries and transport over much larger areas. In particular, experiences with cropland watersheds on relatively level land with highly developed tile drainage do not provide evidence for equivalent changes in nutrient loadings over large river basins with multiple land uses, more variable slopes, and longer river systems. Not only do large-scale nutrient systems tend to be naturally buffered through enrichment, dilution, transformation, and retention processes; human responses to changes in agricultural practices tend to be buffered as well. For example, imposing restraints on the application of fertilizer and manure in targeted areas will cause some reduction of agricultural production in those areas, raise the prices of the affected commodities, and induce farmers to increase production, with associated increases in uses of nutrients elsewhere. These effects can only be estimated with multidisciplinary studies of the large areas.

At present, no single model or modeling package is available to simulate the effects of changes in land cover, land use, and land-management practices on nutrient export and water quality in the Mississippi River Basin as a whole. However, work is underway to develop such capabilities. The HUMUS Project (Hydrologic Unit Modeling of the United States) is an ambitious, continental-scale effort designed to integrate the use of a geographic information system (GIS) for landscape-based input data with hydrologic models of land and water areas in the 2,108 hydrologic cataloging units (HCUs, or watersheds) of the conterminous United States. The HUMUS Project also includes hydraulic simulation of water flows through the streams, lakes, reservoirs, major rivers, and shallow ground-water aquifers of the HCUs. The project includes capabilities to simulate the generation, transformation, and transport of sediments, nitrogen and phosphorus, and some other water quality variables. An Interface Program (IP) has been developed to automate the extraction of model input data from the GIS databases. The IP has routines to automate the display and analysis of model outputs.

Principal sponsors of the HUMUS Project are the Natural Resources Conservation Service (NRCS), Texas Agricultural Experiment Station, and the U.S. Department of Agriculture's (USDA's) Agricultural Research Service (ARS); other local, state, and federal agencies also are supporting the project. Input data have been assembled from many sources, including the USGS, the National Weather Service, the Bureau of the Census, the USDA's Economic Research Service , and the U.S. Army Corps of Engineers.

The GIS database includes digital maps of land cover, land use, crop distribution, topography, soils, hydrography, watershed boundaries, political boundaries, fertilizer uses, manure disposal rates, and crop-management practices for the watersheds in the 48 conterminous states. Most of these maps are scaled at 1:250,000. The database includes historical daily weather records from National Weather Service data for more than 8,000 weather stations for the period 1960–89. Land-use and land-cover data were derived from LANDSAT TM imagery by a USGS-led project (Land Use Data Acquisition, LUDA, circa 1980). Crop-distribution data were derived from the USDA's Census of Agriculture for 1987 and 1992. The soils map is the 1991 STATSGO map from the NRCS. Topographic, hydrographic, and watershed boundary maps were taken from USGS databases. Crop management, fertilizer use, and manure management information was assembled by NRCS from various sources.

The land and hydrologic model used in the HUMUS Project is the “Soil and Water Assessment Tool” (SWAT) (Srinivasan et al. 1993, 1995, 1998; Arnold et al. 1998). The subwatershed area component of SWAT was developed from EPIC, probably the most widely used agricultural crop- and erosion-simulation model available. The EPIC component of SWAT simulates the growth and production of each kind of vegetation in each HCU through daily time steps of temperature, rainfall, other weather conditions, tillage practices, nutrient applications, and harvest operations. Soil, plant, and water interactions are modeled for as many as 10 layers of the selected representative soil profiles associated with each kind of vegetation. In this component of the model, nutrient and sediment deliveries are simulated to the edges of fields and bottoms of root zones. Because of a lack of assembled data, nitrate and dissolved P traveling through the ground waters are assumed not to be transformed in this oxygen-limited environment below the root zone until they discharge into surface-water systems.

The hydraulic components of SWAT are designed to “collect” water from land areas and route it through a watershed’s streams, lakes, reservoirs, and rivers. Surface-water runoff is combined with ground-water return flows from lateral flow through the root zone and from shallow ground-water aquifers. Stream-flow velocities are estimated from slope, channel shape, and roughness characteristics. Ground-water return flow rates (lag times) are estimated from the analysis of recession hydrographs derived from representative stream-gauge records. The effects of tile drains can be simulated by reducing the lag time for ground-water return flows. A generalized reservoir inflow–outflow relationship has been developed to simulate changes in lake and reservoir storage through time. EPA’s QUAL2E model also has been incorporated into SWAT to simulate the fate and transport of nutrients in aquatic environments. The QUAL2E component of the model tracks such variables as temperature, dissolved oxygen, nutrient assimilation, nitrification, denitrification, and P cycling through aquatic systems.

Results of the simulation modeling are being correlated with flow and water quality data at hundreds of stream-gauging stations in those states. Project results also are being compared with results of studies conducted by the USGS and the National Oceanic and Atmospheric Administration (NOAA). Good correlations of average annual simulated stream flows with gauged stream-flow records have been achieved for most of the major river basins east of the Rocky Mountains and in many of the watersheds in the intermountain and Pacific basins.

For a variety of reasons, including limited data and inherent inaccuracies in the water quality monitoring records available for comparing with the results of simulations, correlations of model outputs for N and P concentrations and loads with measured values are more tentative. Some factors that make it difficult to compare the HUMUS Project nutrient simulations with water quality monitoring data are: (1) incomplete information about the locations and effects of drainage systems, local wetlands, and major confined animal production facilities; (2) difficulties associated with including point-source nutrient information in the model on a daily basis at the hydrologic cataloging unit (HCU) scale; (3) the fact that water quality monitoring sites and point-source discharge sites usually are not located near each other or near the outlets of the eight-digit HCU areas; and (4) the specific locations and effects of large, confined, animal production sites, tile-drainage systems, and local wetlands are not generally available in the database. Point-source nutrient discharge data, for example, are provided only as estimates of average annual discharges of total N and P. They do not provide estimates of how much of each nutrient is in the organic or inorganic form at the point of discharge or how these discharges change on a daily or seasonal basis. Nevertheless, simulations of concentrations and loadings of N and P for the major rivers in the Mississippi River Basin have produced estimates on the same order of magnitude as those in the monitored records for these nutrients, and they compare well with estimates of such loadings for HCUs in the basin made by the USGS with its SPARROW Project. The accuracies of SWAT simulations of nutrient concentrations and loads will improve significantly as the HUMUS Project continues to be developed and improved.

Using the HUMUS Project/SWAT model approach and geographic, soils, crop, and management databases developed for the HUMUS Project, Arnold, Srinivasan, and Walker (Blackland Research Center, Temple, TX, unpublished data) estimated potential changes in nutrient concentrations and flows in local watersheds and the major rivers of the Corn Belt. They based crop nutrient management on the concept of a “percent of biomass requirement.” Specifically, N fertilizer application rates were allowed to range from 85% to 130% of the amounts of nutrients needed from fertilizer application to meet optimum biomass production levels, after the sources of those nutrients available from the soil and residue carryover were considered. Figure 3-15 shows results of running the HUMUS/SWAT model to estimate total N outflows for four simulated levels of N inputs for the areas above the Mississippi River at Alton and the Ohio River at Cannelton Dam and near Metropolis. The figure shows estimated ratios of expected loads, compared with current average annual flows of total N at these locations. The ratios were derived by assuming that the N outflows associated with the assumed input level of 115% of plant requirements correspond to the current average level of uses of N fertilizers by farmers in the region.

In an independent approach, Benson and Atwood (NRCS, Temple, TX, unpublished report) used EPIC and the Agricultural Sector Model (ASM) to estimate the farm-, regional-, and national-level physical and economic impacts of imposing N fertilizer restraints in the Corn Belt. The analyses are based on simulations of corn, soybean, and sorghum production for ~1,500 field sites in parts of Illinois, Indiana, Iowa, Minnesota, Missouri, Nebraska, Ohio, and Wisconsin within the MRB. Current conditions were estimated based on specific land-use and land-management data for these sites in the *1992 Natural Resources Inventory* and state and regional information on fertilizer and crop management practices assembled from various sources by Benson and others since 1993. Their analysis of nutrient management alternatives was based on a stress concept that proposes to require row-crop farmers to reduce N fertilizer applications to levels that would result in corn, corn–soybean, and sorghum crops being in N stress for 5–10% of the average growing season. Table 3.3 summarizes Benson and Atwood’s findings.

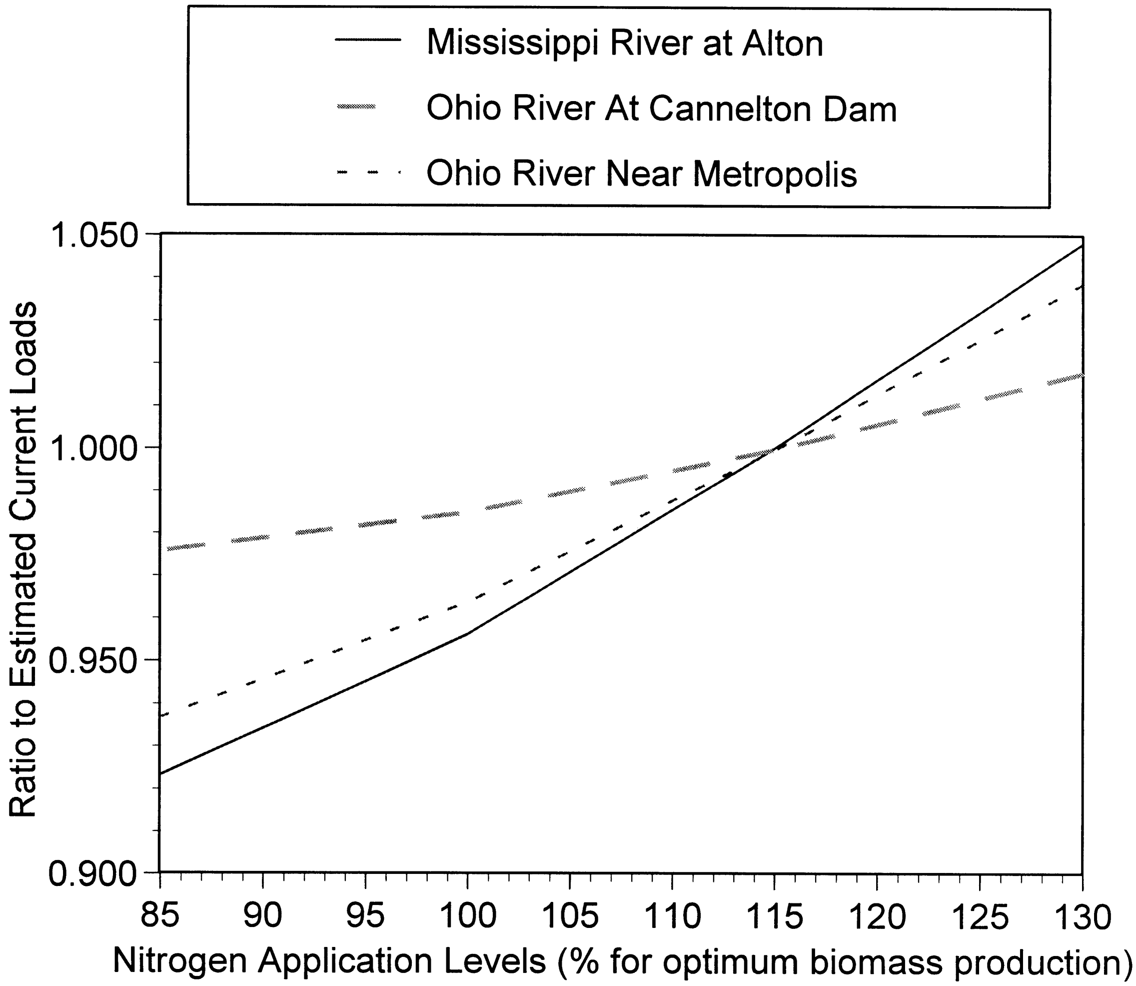


FIGURE 3.15. Changes in nitrogen load in downstream rivers expected from changing application rates of fertilizer nitrogen. NOTE: Estimated current situation is at X = 115. Estimated application level corresponding to having corn, sorghum, and corn–soybean crops in nitrogen stress about 10% of an average season occurs at X = 98. (*From Arnold, Atwood, Benson, Srinivasan, and Walker, unpublished, Blackland Research Center, Temple, TX, 1998*.)

The two studies thus used different approaches and are based on somewhat different data and concepts. A general comparison of the estimates of nitrate discharges from the two approaches suggests that the baseline (*current average*) condition for the Benson–Atwood approach corresponds roughly to the 115% N input level in the HUMUS Project approach, and that the 10% stress level in the Benson–Atwood approach corresponds roughly to a 98% of biomass demand level in the HUMUS Project approach.

Comparison and analysis of the results of the two studies, including the results presented in Table 3.3 and Figure 3.15, lead to the following tentative conclusions regarding fertilizer management options for reducing loads of nutrients discharged from the Corn Belt to the Mississippi River. Because the HUMUS/SWAT modeling approaches are still under development and the model output has not been verified by actual measurements, the conclusions should be regarded as hypotheses.

TABLE 3.3. Potential effects of imposing fertilizer-use reductions based on plant nutrient stress in the Midwest on crop yields, losses of N and P, erosion rates, and various economic conditions.

Nitrogen Nutrient Stress Target[1]	**Time in Stress** *5%*	*10%*
	Related Changes	
Change in nitrogen applied in fertilizer:		
On continuous corn	-17.0%	-27.0%
On corn–soybean rotation	-30.0%	-40.0%
On continuous sorghum	-19.0%	-32.0%
Combined area weighted effect	-23.9%	-34.0%
Change in crop yield:		
On continuous corn	-3.3%	-8.8%
On corn in corn–soybean rotation	-2.1%	-4.3%
On soybeans in corn–soybean rotation	-0.4%	-0.4%
On continuous sorghum	-3.5%	-8.6%
Change in total nitrogen discharges from fields:		
Of continuous corn	-9.2%	-13.3%
Of corn and soybeans in rotation	-3.5%	-4.7%
Of sorghum	-15.3%	-24.5%
Combined area weighted effect	-5.4%	-7.7%
Change in organic phosphorus discharges from fields:		
Of continuous corn	-1.9%	-0.5%
Of corn and soybeans in rotation	0.0%	0.0%
Of sorghum	1.6%	2.7%
Combined area weighted effect	-0.5%	-0.1%
Change in sheet and rill erosion rates due to changes in crop patterns caused by imposing nitrogen stress-based restrictions	-0.2%	-2.9%
Overall change in crop producers' welfare in the U.S. (due to fertilizer savings and crop price increases)	1.69%	3.63%
Change in crop producers' welfare in treated area in the Corn Belt	1.71%	3.42%
Percent of farmers in study area who would benefit from imposing nitrogen stress-based fertilizer-use restrictions	44%	59%
Percent of farmers in study area who would lose profits due to imposing nitrogen stress-based restrictions	56%	41%
Change in the portion of U.S. consumers' welfare associated with consumption of agricultural sector products	-0.04%	-0.12%

[1]*Average percent of growing season that selected crops are to be in nitrogen stress.*

NOTE: Estimates of changes in N and P discharges are for direct losses from crop fields, not from transport to rivers, streams, and lakes. Percent changes are based on estimates of current fertilizer use and management on 1992 crop acreages.

Source: Benson and Atwood, NRCS, Temple, TX, unpublished report, 1998.

- Nitrogen fertilizer applications in Corn Belt states could be reduced by as much as 30% on continuous corn and sorghum and by as much as 40% on corn and soybean rotations without seriously reducing the *national* production of these crops. Total national production of corn would be reduced by less than 1.5%, and soybean production by less than 0.6%. Sorghum production would increase by up to 2.2%.
- If all corn, sorghum, and soybean farmers in the Corn Belt were required to reduce their N fertilizer applications to levels that would impose N stress on those crops (based on having the crops in N stress about 10% of an average season or on applying only ~98% of the N needed for optimum average season biomass production), almost all of them would have reduced crop yields. Nevertheless, about half of those farmers would have increased profits because of higher crop prices and lower fertilizer costs; the other half would lose profits in spite of these factors. If this strategy were adapted by or imposed on all farmers in the Corn Belt, the amount of N discharged from all croplands in the Corn Belt could be reduced by as much as 7.7%.
- The amount of phosphorus being discharged from croplands would not be changed significantly by a program that focused only on reducing amounts of fertilizer applied to crops. Other practices, such as treating point sources, reducing erosion rates, and controlling sediments would be required if P discharges to streams were to be reduced significantly.
- A policy that would impose reductions in applications of N fertilizers on corn, corn–soybean, and sorghum fields in the Corn Belt by about 34% would decrease the total amount of N transported from the Upper Mississippi and Ohio Rivers by 2–5%.

There are many reasons why the results from these two studies may not directly correspond to the results of other studies, especially studies of specific local sites and watersheds. One reason is that many local studies are based on experiments on single crops and relatively small areas. For example, relative to the TN discharges in Table 3.3, it can be seen that a 24% reduction in N deliveries from all sorghum fields and a 13% reduction in N deliveries from all corn fields would result in less than an 8% reduction in deliveries from all of the cropland fields, partly because of the benefits of having large portions of the cropland in the region in soybeans. The actual reductions in deliveries from the total area would be expected to be even less, because there are significant areas of noncropland that receive no fertilizer applications.

A second reason for differences is that some studies, notably those on the Minnesota River Basin, involve areas that have significant effects from tile drainage. These effects probably are not fully accounted for in the HUMUS Project or in simulations with EPIC. A third reason for differences may be related to the local effects of point-source discharges.

An example of the likely importance of tile drainage may be found in the fact that simulation results for the Minnesota River Basin in the HUMUS Project predict a load of only about 7,000 metric tonnes of nitrate-N a year, but water quality monitoring data for the river at Jordan, MN, indicate that the average annual load of nitrate-N may be as high as 47,000 tonnes a year. Because a very high percentage of the cropland in the basin is tile-drained, this difference suggests that there is a significant opportunity to reduce nitrate loading from the basin by controlling and/or capturing the discharges from the tile drains. Similar comparisons may suggest other areas where nutrient capture and/or assimilation efforts could be useful in reducing nutrient deliveries downstream.

3.2.5.5 NUTRIENT RETENTION WITHIN THE RIVERS OF THE MRB

Because nutrients are highly reactive substances in both terrestrial and aquatic ecosystems, they tend to behave very nonconservatively there. For example, a variety of loss and retention mechanisms (e.g., assimilation by plants and deposition into bottom sediments) tends to make outputs of N and P from a given aquatic ecosystem (e.g., a lake or stretch of river) lower than the inputs to the ecosystem.

The possibility that internal sources (e.g., biological N_2 fixation and release of N and P from bottom sediments) may be important leads to the possibility that outputs may exceed inputs under certain circumstances. Thus, predicting the effects of reducing nutrient inputs to the Mississippi River and its tributaries on the outflow of nutrients to the Gulf of Mexico is difficult because the extent to which retention and internal losses or gains occur must be taken into account. This is especially important for nutrient-source reductions in the Upper MRB because of the long distances and travel times (a scale of several thousand kilometers) before river water from these areas reaches the Gulf. At the same time, it must be noted that few measurements are available to directly assess the extent of nutrient retention in MRB rivers. Consequently, it was necessary to approach this question indirectly by examining evidence from modeling studies and intra-seasonal analyses of nutrient concentrations at selected sites in the system.

Mechanisms of nutrient retention in rivers

Although little is known about nutrient removal processes in the Mississippi River and its tributaries, information from the literature provides insight as to the most important processes and their possible effects on nutrient transport in the aquatic systems of the Mississippi watershed.

Phosphorus enters streams in both dissolved and particulate forms, but is most commonly transported in particulate forms. Dissolved phosphorus is converted rapidly to particulate forms by phytoplankton uptake in the water column or by adsorption to fine silts and clays suspended in the water column (Ryden et al. 1973; McCallister and Logan 1978). Dissolved phosphorus also can be removed from the water column directly by aquatic macrophytes and by periphyton growing on rocks and bottom deposits. Particulate phosphorus is removed from the water column by settling and is deposited in the bottom sediment when the velocity of flowing water is insufficient to keep the particles in suspension. Sedimentation processes are especially important under low-flow conditions, in slow-moving parts of rivers (where the downstream vertical gradient is small and the river channel widens), and in backwater pools of larger rivers and reservoirs.

Benthic denitrification is believed to be the principal process governing the permanent removal of nitrogen from rivers and streams (Seitzinger 1988, Howarth et al. 1996). However, uptake by macrophytes and deposition of particulate organic N also may be important in some parts of the MRB, particularly in the backwater areas and navigation pools formed by the series of 30 lock-and-dam structures in the Upper Mississippi River. Significant N losses by sediment accumulation also are likely in the many large impoundments that have been constructed on major tributaries in the MRB (e.g., especially in the Tennessee and Missouri Rivers). Long-term physical retention of particulate N in flood plains may account for additional N losses in some aquatic systems (Johnston et al. 1984; Billen et al. 1989), including rivers of the MRB. Estimates of N losses in rivers vary from < 5% to as much as 80% of the external N inputs.

Denitrification is a biologically mediated process that involves the reduction of nitrate to gaseous nitrogen forms (primarily N_2 but also some N_2O and traces of NO and NO_2). Denitrification occurs under anoxic conditions in benthic sediments. Two mechanisms have been proposed to supply nitrate to the anoxic sediment zone: direct diffusion of nitrate from the water column (e.g., Seitzinger 1988; Howarth et al. 1996; Kelly et al. 1987; Baker and Brezonik 1988) and nitrification of ammonium in the surficial aerobic sediment layer. This ammonium is supplied by mineralization of organic N in the aerobic sediment layer; in turn, the organic N originates from primary production in the stream (e.g., Seitzinger 1988; Novotny and Olem 1994).

Several studies have demonstrated the influence of various chemical and physical properties on stream denitrification and on aggregate measures of N loss (e.g., Seitzinger 1988; Howarth et al. 1996; Kelly et al.

1987; Behrendt 1996; Bachmann et al. 1991). These factors include oxygen concentrations in the water column, organic content of sediments, channel depth, water residence time, and runoff. However, the effects of these factors on N loss rarely have been examined systematically over a range of river sizes and watershed scales. Channel depth, an important limiting factor, influences denitrification rates in stream sediments in several ways. First, other factors being equal, increasing the water-column (channel) depth decreases the fraction of nitrate in the water that can diffuse across a given area of the sediment–water interface in a given period of time (e.g., see Howarth et al. 1996; Huang and Wozniak 1981). Second, channel depth affects settling rates and settling times of particulate organic N. Third, channel depth affects the relative importance of the benthic layer versus the water column as the site for photosynthetic production of organic matter in streams.

Regional analysis of nutrient losses in MRB rivers and streams

Based on a modeling analysis of stream-monitoring data, R.A. Smith et al. (1997) recently examined the importance of channel size on stream nutrient loss and provided a systematic description of how total N and P losses vary in U.S. rivers. In-stream losses of nutrients (TN and TP) were observed to vary inversely with channel size according to a first-order decay process. The empirical rates of nutrient loss accounted for the mean water residence time in channels and varied with channel size by approximately an order of magnitude. This study provides a basis to estimate the nature of nutrient losses in the aquatic systems of the Mississippi River Basin and is applied in a simple analysis of regional-scale nutrient losses in rivers of the basin. An extension of this study to a detailed analysis of N losses in the MRB and their effect on the quantity of nitrogen delivered to the Gulf of Mexico currently is being conducted by the USGS.

In the analysis conducted for this report, in-stream losses of nutrients (TN and TP) were estimated for the Upper Mississippi (including the Ohio and Tennessee watersheds and excluding the Missouri) and the Lower Mississippi based on (1) first-order nutrient loss rates (R.A. Smith et al. 1997) and (2) estimates of mean water travel times in the regional watersheds derived from a 1:500,000-scale drainage network of U.S. rivers. Estimates of in-stream losses of nitrogen were made for small tributaries and large rivers classified according to a mean flow rate below and above 10,000 ft^3/s, respectively. This classification separates the mainstems of the major rivers—including the Ohio, Wabash, Cumberland, Tennessee, Illinois, Arkansas, White (in both Indiana and Arkansas), Yazoo, and Mississippi—from their smaller tributaries.

The results (Table 3.4) show that although the mean water travel times in the small tributaries are about one-half to two-thirds of the mean travel times of the larger rivers (mean travel times of 2.6–3.7 days versus 4.9–5.7 days, respectively), estimated nutrient losses in small tributaries are more than twice as high as those in the mainstem rivers. This reflects the effect of higher nutrient loss rates in rivers with mean flow < 10,000 ft^3/s. These (small) rivers are estimated to have first-order loss rates > ~10% per day of travel time (R.A. Smith et al. 1997).

In Table 3.4, we express the quantity of nutrient flux removed by in-stream processes on a regional basis as the mean of the in-stream nutrient mass removed from all tributary and mainstem reaches in a region. The mean percentage loss of TN is estimated to range from 35% to 40% in the small tributaries of the Upper and Lower Mississippi regions. In comparison, the TN loss is about 20% in the larger mainstem river channels in these regions. The mean percentage loss of TP, which ranges from 28% to 37% in the small tributaries, compares with negligible losses estimated for the mainstem channels. Comparison of the regional results also indicates that the mean percentage loss of TN and TP in the small tributaries of the Upper Mississippi is about 1.3 times that estimated for the tributaries of rivers in the Lower Mississippi region. In the mainstems of the larger rivers of the Upper and Lower Mississippi regions, similar percentage losses are estimated for nutrients.

TABLE 3.4. Estimated mean water travel times and percentages of in-stream nutrient removal/retention by in-channel processes in tributaries and mainstem rivers of the Upper and Lower Mississippi River Basins.[1]

Region	Water Travel Days		% TN Loss		% TP Loss[2]
	Tributaries	Mainstem	Tributaries	Mainstem	Tributaries
Upper Mississippi	3.7	5.7	45	20	37
Lower Mississippi	2.6	4.9	35	18	28

[1]The Upper Mississippi includes rivers of the Ohio, Tennessee, and Upper Mississippi River water resource regions (see Figure 3.22). Tributaries and mainstem rivers were defined as rivers with a mean flow rate below and above 10,000 ft^3 sec^{-1}, respectively. Reaches were defined according to a 1:500,000-scale drainage network of U.S. rivers (R.A. Smith et al. 1997).
[2]TP loss in mainstems was estimated to be negligible (R.A. Smith et al. 1997).

Evidence for nutrient retention from seasonal variability in concentrations

Nutrients in the Mississippi River system may change in form or quantity seasonally because of activities of river plankton (primarily algae, but also bacteria and other microbes). Water quality data from the Minnesota River at Jordan, MN, were used to examine relationships between river phytoplankton and nutrients. Preliminary inspection of the long-term data from 1980 to 1992 suggested that minimum annual chlorophyll *a* (chl *a*) values generally occur during May, and maximum values usually occur in September.

Based on this tentative finding, we plotted monthly average chl *a* concentrations for May of each year, versus corresponding averages for September (Figure 3.16). In general support of this inference, most data points fall above the 1:1 line. In contrast, total nitrogen (TN) and nitrate have higher levels at this site in spring than in summer, but that probably reflects seasonal trends in export of nitrogen from the landscape (the highest values would be expected during spring runoff). The bulk of TN in the river at this site is nitrate; organic N and ammonium concentrations do not show strong differences between spring and summer and typically have much lower concentrations than nitrate. In contrast, higher concentrations of total phosphorus (TP) and soluble reactive phosphate typically occur at this site in summer than in spring. This is surprising, given that there is a strong sediment-associated component of TP that is transported during the high river flows of spring. Perhaps P is diffusing out of the river-bottom sediments under low-oxygen conditions in the bottom water of the river during the warm summer months.

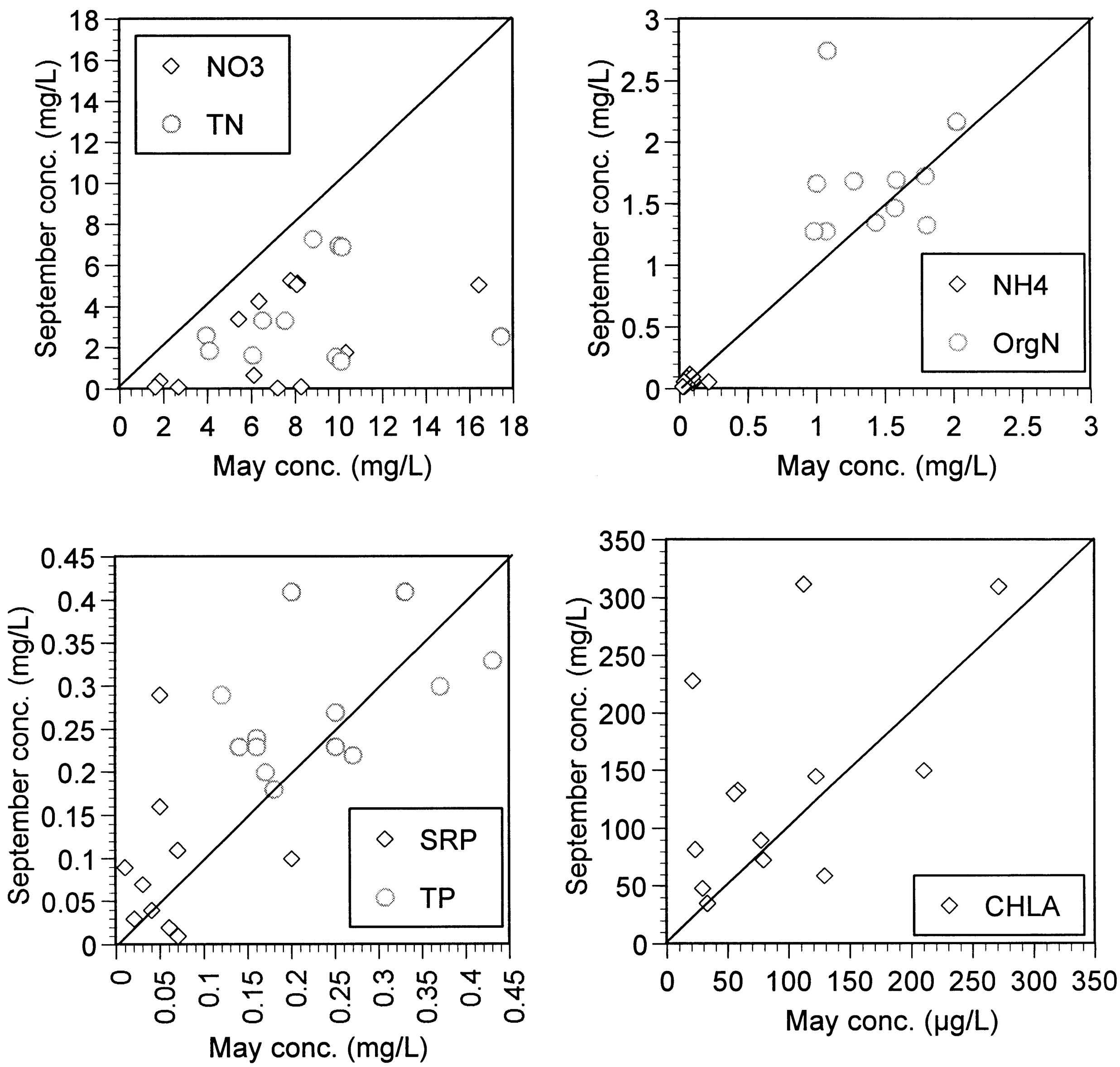


FIGURE 3.16. Plots of late summer (September) concentrations of various nutrient (N and P) forms and chlorophyll *a*, versus corresponding spring (May) values for the Minnesota River at Jordan, MN. NOTE: Each datum represents the mean monthly values for a given year between 1980 and 1992. Not all years had data for all variables; thus, 13 data points are not present on all graphs. (*Data from Metropolitan Council Environmental Services, St. Paul, MN.*)

Additional insight into nutrient variability can be seen by examining the monthly means during a given year. Figure 3.17 plots nutrient and chl *a* concentrations at the Jordan site during 1990, a year in which seasonal sampling was unusually detailed. Two annual peaks were found for chl *a*—one in late winter/early spring and the other in late summer. In contrast, organic and inorganic suspended sediment and dissolved inorganic nutrient concentrations peaked during late spring/early summer, when chl *a* levels were at a minimum. These trends suggest seasonal variations in nutrient processing within the river. However, it is not clear whether inorganic nutrient concentrations were low during January–April and September–December because of uptake by phytoplankton (i.e., internal cycling processes) or simply because of low rates of supply from suspended sediments (which were low in concentration) and from runoff from the landscape. Further investigations are needed to delineate the relative importance of these explanations.

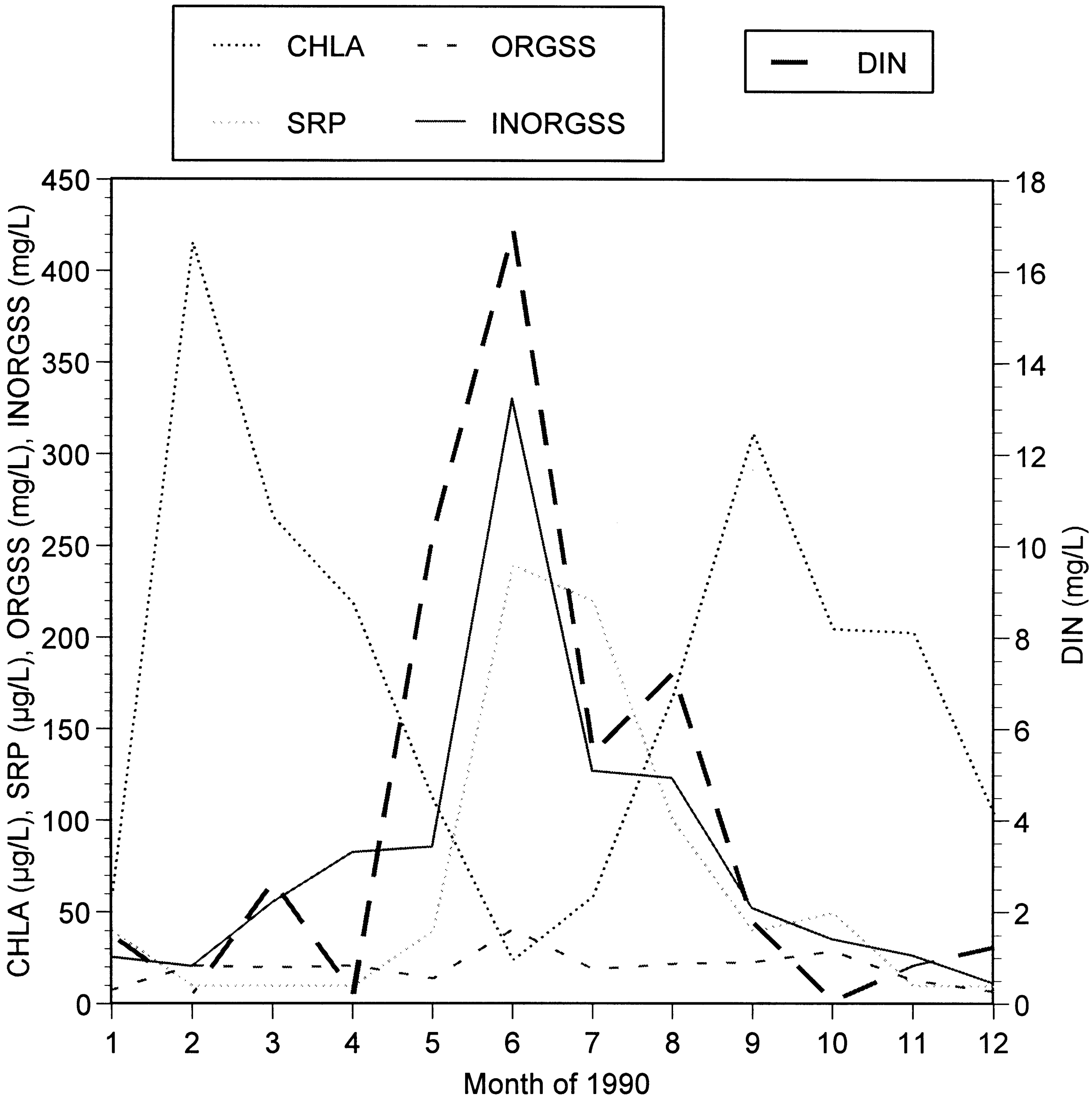


FIGURE 3.17. Mean monthly concentrations of chlorophyll *a*, organic and inorganic (nonvolatile) suspended solids, and dissolved inorganic N (sum of nitrate-N and ammonium-N) for the Minnesota River at Jordan, MN, during 1990. NOTE: Sampling frequency was 2–4 times per month. (*Data from C. Larson, Metropolitan Council Environmental Services, St. Paul, MN, 1998.*)

In a related effort to separate the effects of internal cycling from variations in external loadings to the river on seasonal variations in nutrients, we examined concentrations of N forms along longitudinal transects in the Upper Mississippi River (Figure 3.18) and Minnesota River (Figure 3.19). In both cases, the plots show downstream trends in relative composition of the N forms (upper panels) and downstream trends in actual concentrations of the N forms (lower panels). Comparing longitudinal patterns for the same river stretch between two seasons tends to minimize the influence of external loading (i.e., inputs from runoff) on the spatial pattern. Our hypothesis was that if internal processing were important, we would see relatively greater decreases in concentrations (or smaller increases in concentrations) in a downstream direction during the warm summer period than during the cold spring period.

Bukaveckas et al. (1998) recently used this type of analysis on a stretch of the Ohio River and reported some evidence for seasonal nutrient processing/retention. The trends shown in Figures 3.18 and 3.19 do not provide strong support for this hypothesis. However, it would be premature to reject the hypothesis based on this preliminary effort. Indeed, routinely monitored stations—the only sites on the rivers having data for such an analysis—generally are not located ideally for such analysis. Further studies should be conducted using shorter longitudinal transects selected to avoid the confounding influences of tributary inputs. Measurements of N species conversion rates also are needed to address the temporal component of N cycling.

The presence of ~30 locks and dams along the mainstem of the Upper Mississippi River (UMR) above St. Louis creates pools in the river that can behave like reservoirs. Studies are underway to examine the mass-balance behavior of nitrogen in Pools 8 (above La Crosse, WI) and 13 (south of Dubuque, IA) by the lock-and-dam system. Preliminary results for Pool 8 (Soballe 1998) indicate that nitrate and TN outputs exceed inputs during high flows, but outputs are similar to inputs during normal to low flows. Ammonium concentrations are lower during the summer and fall compared to the winter and spring. In contrast, nitrate inputs to Pool 13 exceed outputs throughout the year. Ammonium inputs exceed outputs during low-flow periods, but inputs are less than outputs during higher flows. Ammonium concentrations tend to increase during high flows. TN loads are strongly related to flow, and inputs are typically balanced by outputs. Based on these preliminary results, it appears that the lock-and-dam system in the UMR does not exert a major, long-term effect on N retention in the UMR, but further studies are needed to verify this statement.

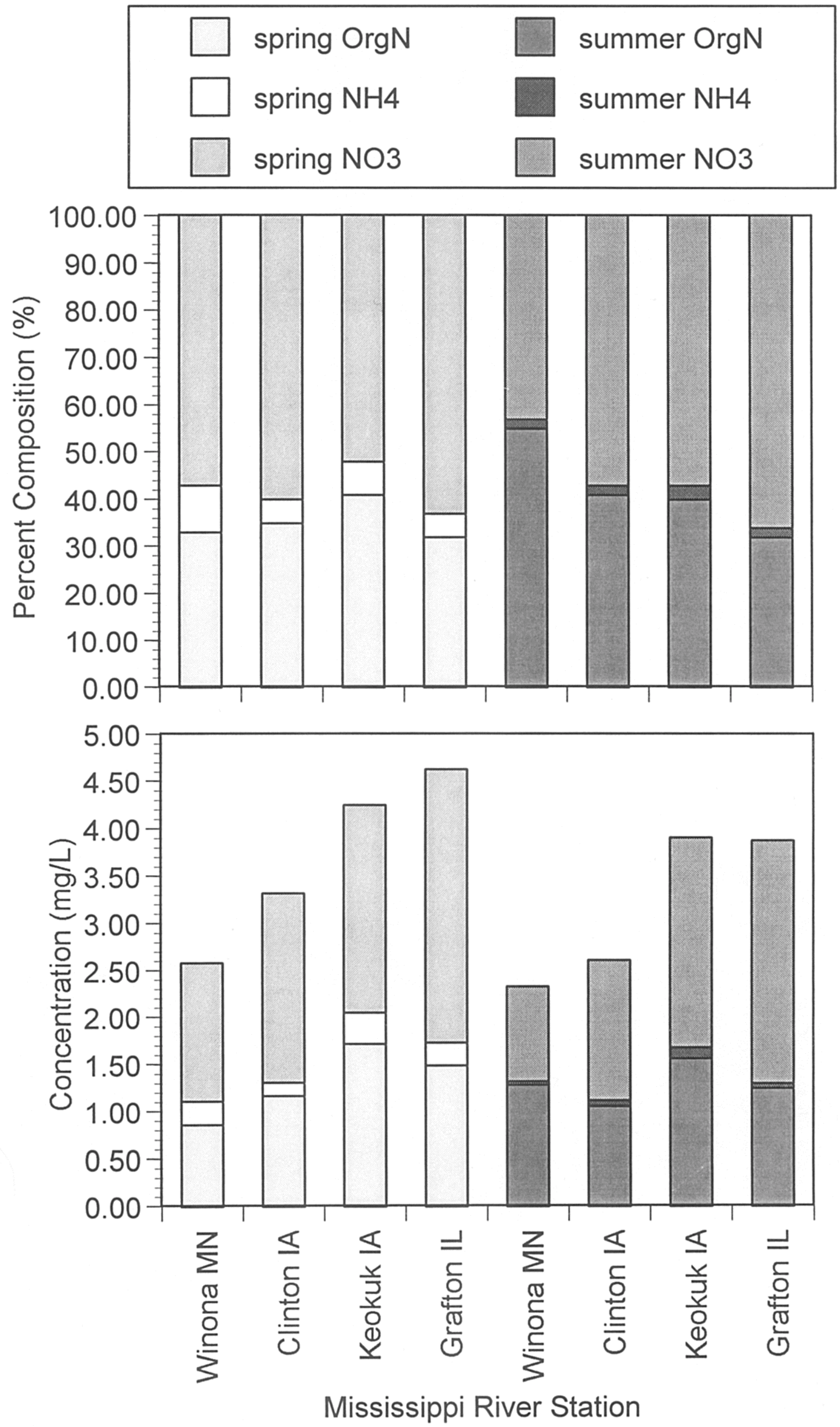


FIGURE 3.18. Percent composition (upper panel) and concentrations (lower panel) of nitrogen forms in spring (four bars to the left) and summer (four bars to the right) for four stations in the Upper Mississippi River. NOTE: Stations are plotted left to right in order of downstream direction. (*Data from D. Goolsby, USGS, Denver, Co, personal communication, 1998*.)

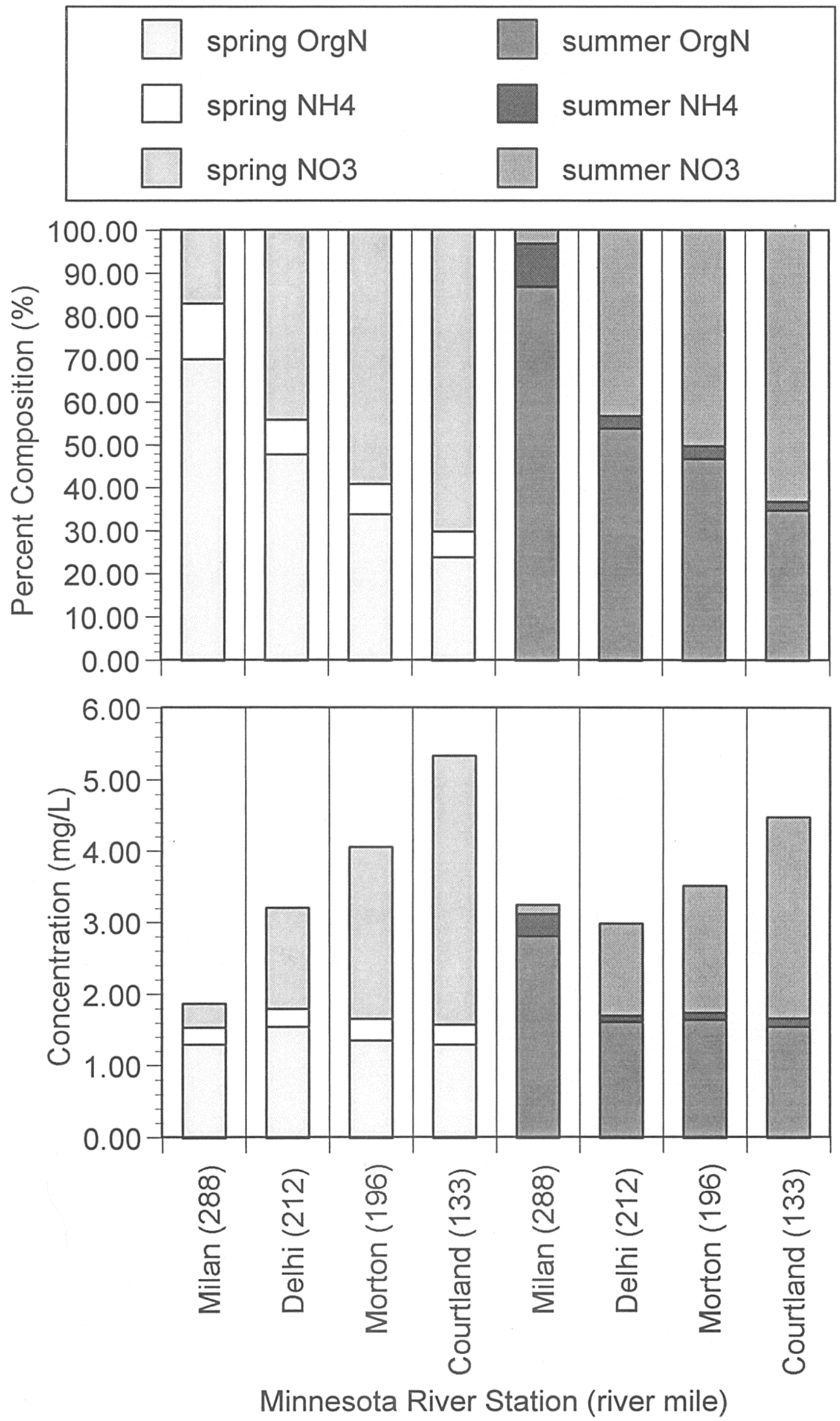


FIGURE 3.19. Percent composition (upper panel) and concentrations (lower panel) of nitrogen forms in spring (four bars to the left) and summer (four bars to the right) for four stations in the Minnesota River. NOTE: Stations are plotted left to right in order of downstream direction. (*Data from various agencies and assembled by D. Mulla, University of Minnesota, St. Paul, MN.)*

3.2.6 Effects of Nutrient Changes on the MRB Aquatic Ecosystem

3.2.6.1 DECREASED INCIDENCE OF VIOLATIONS OF WATER QUALITY STANDARDS

We addressed the potential for decreased frequency of violations of water quality standards in the Mississippi River Basin primarily by examining the current frequency of violations and then extrapolating qualitatively for the frequency of violations if nutrient-source controls were implemented. We used three sources of data to analyze current violations: the 74-station MRB data set assembled by the USGS, the Minnesota River Basin database, and information contained in 1996 state 305(b) reports and the national *1996 305(b) Report to Congress* (USEPA 1998a).

Analysis of basinwide database

The USGS data set on the MRB includes 74 sites across the Mississippi River Basin (Table 3.1) and covers a period from approximately 1973 to 1995. The database was sorted to extract results for dissolved oxygen, nitrate, pH, and un-ionized ammonia that exceeded applicable state water quality standards. These variables were selected because they are the only ones affected by changes in nutrient levels for which legal water quality standards exist. A dissolved oxygen (DO) criterion of 5 mg/L was selected because this concentration reflects the standard used by most states. We selected two values for pH criteria: $\leq$ 6.5 units and $\geq$ 9.5 units; again, most state standards are based on these values.

At present, there are no legal standards for chemical substances that are related specifically to the role of the substances as *plant nutrients*. There are standards for nitrate related to human health effects and for un-ionized ammonia related to its toxicity to aquatic organisms. The drinking-water standard for nitrate, 10 mg N/L, was used for this analysis; strictly speaking, this standard applies only to waters that are classified for use as potable water. Un-ionized ammonia (NH_3–N) violations were calculated according to recent EPA recommendations (USEPA 1998b), using both one-hour criterion maximum concentrations (CMCs, both for salmonids present and absent) and 30-day average criterion continuous concentrations (CCCs).

The results presented in Table 3.5 indicate that violations are uncommon at the MRB sites represented in the database. For example, most stations have a 2% or lower level of noncompliance with the DO standard. Stations with higher frequencies of exceedances, which may indicate the presence of recurring DO problems, include the Grand River at Sumner, MO (7% noncompliance); the Missouri River at Hermann, MO (8%); the Mississippi River at Thebes, IL (6%); the Osage River at St. Thomas, MO (9%); the Ouachita River at Columbia, LA (15%); the Red River above Simmesport, LA (6%); and the Tennessee River at Paducah, KY (13%).

Most stations also had a 2% or lower level of noncompliance with pH standards. The lower pH standard is possibly being exceeded in the Allegheny River at New Kensington, PA (12% noncompliance); the Big Black River at Melville, LA (12%); the Monongahela River at Braddock, PA (6%); the Ohio River at Wheeling, WV (11%); the Ouachita River at Columbia, LA (42%); and the Tennessee River at Pickwick Landing Dam, TN (13%). Some of these violations, particularly in the eastern rivers, where softer (low-alkalinity) waters are common, may reflect natural conditions. The pH 9.5 criterion level was rarely reached; 6 of 250 (2.4%) samples of the Rock River at Joslin, IL, exceeded this standard.

TABLE 3.5. Violations of water quality standards over 1973–93 for nutrient-related variables in 74 Mississippi River Basin sampling stations.[1]

Station ID #	Percent Violations				Station ID #	Percent Violations			
	DO	pH	NO_3	CCC[2]		DO	pH	NO_3	CCC[2]
Ohio River System					**Missouri River System**				
3049625	1.4	12.1	0.0	0.0	6294700	0.0	0.0	0.0	1.7
3438220	0.0	3.5	0.0	0.0	6439300	0.0	0.0	0.0	0.0
3274600	1.3	0.0	2.2	2.5	6902000	6.8	0.0	0.0	0.0
3321230	2.5	6.4	0.0	0.0	6478500	4.1	0.6	0.0	0.0
3201300	2.5	2.9	0.0	0.0	6892350	0.0	0.0	0.0	0.8
3290500	0.0	2.6	0.0	0.0	6174500	0.0	0.0	0.0	0.0
3085000	2.0	6.5	0.0	0.0	6338490	0.0	0.0	0.0	0.0
3150000	3.6	0.5	0.0	0.0	6440000	0.0	0.0	0.0	0.0
3303280	0.0	1.0	0.0	0.0	6090800	0.0	0.0	0.0	0.0
3216600	0.0	6.8	0.0	0.0	6453000	0.0	0.0	0.0	0.0
3277200	6.7	3.8	0.0	0.0	6934500	7.7	0.0	0.0	0.0
3112510	0.0	3.6	0.0	0.0	6109500	0.0	0.0	0.0	0.0
3612500	2.1	11.1	0.0	0.0	6132000	0.0	0.0	0.0	1.2
3234500	3.7	0.0	0.0	0.0	6185500	0.0	0.0	0.0	0.0
3609750	13.3	4.4	0.0	0.0	6115200	0.0	0.0	0.0	0.0
3593005	3.0	12.7	0.0	0.0	6926510	8.9	0.0	0.0	0.0
3571850	5.3	3.1	0.0	0.0	6805500	0.7	2.3	0.0	0.0
Mississippi River Mainstem					6295000	0.0	0.0	0.0	0.0
7374508	0.0	0.0	0.0	0.0	6329500	0.3	0.0	0.0	0.9
7374525	0.7	0.0	0.0	0.0	6296120	0.0	0.0	0.0	0.0
5420500	4.4	0.0	0.0	0.0	**Southern Plains System**				
5474500	4.2	0.0	0.0	2.6	7263620	0.0	0.9	0.0	0.0
7032000	1.6	0.0	0.0	0.0	7152500	0.0	0.3	0.0	0.0
5331570	1.0	0.0	1.1	8.0	7164500	0.0	1.2	0.0	0.0
5331000	2.4	0.0	1.7	0.0	7231500	0.4	0.3	0.0	0.8
7022000	6.4	0.0	0.0	0.0	7245000	0.7	0.4	0.0	0.0
7289000	0.0	2.0	0.0	0.0	7355500	1.4	1.4	0.0	0.0
5378500	0.0	0.0	0.0	0.0	7344410	0.4	0.0	0.0	0.0
5587455	1.7	0.0	0.6	0.6	7337000	0.0	0.0	0.0	0.0
5267000	1.9	0.0	0.0	0.0	7355601	6.4	8.9	0.0	0.0
7373420	0.0	0.3	0.0	0.0	7047900	2.5	1.6	0.0	0.0
Upper Mississippi River System					7077800	0.0	0.0	0.0	0.0
5369500	0.0	2.0	0.0	0.0	**Lower Mississippi River System**				
5543500	1.3	0.0	0.0	14.0	5594100	2.2	2.1	0.0	0.0
5586100	4.4	0.3	0.0	3.4	7381495	1.6	0.5	0.0	0.0
5465500	0.0	0.0	2.8	1.9	7290000	1.2	12.3	0.0	0.0
5330000	1.9	0.0	7.7	0.5	7381600	3.2	0.7	0.0	0.0
5446500	na	0.0	0.0	2.7	7367640	14.9	41.9	0.0	0.0
5340500	0.0	1.0	0.0	0.0					
5407000	1.5	0.0	0.0	0.0					

[1] DO violations as ≥ 5 mg/L; pH violations as ≥ 6.5 units; and nitrate violations as ≥ 10 mg/L NO_3-N. The only stations with violations as ≥ 9.5 pH units were 5586100 (0.3%), 6453000 (0.9%), 5420500 (1.1%), 6805500 (0.3%), and 5446500 (2.4%).

[2] One-hour average acute values (CMC: Criterion Maximum Concentration) for un-ionized ammonia (both with and without salmonids present) had no violations at any station. CCC (Criterion Continuos Concentration) values presented are not calculated 30-day averages, but represent instantaneous values violating

the 30-day criterion and serve only as a warning of potential problems at a given station (see U.S. EPA 1998b).

Almost all of the examined stations (68 of 74) had no samples that were noncompliant with the drinking-water standard for nitrate. There are possible nitrate problems for the Minnesota River at Jordan, MN (8% noncompliance). Of course, nitrate concentrations far lower than the drinking-water standard are a cause for concern relative to eutrophication problems (see sections 3.2.6.3 and 3.2.6.4), but no legal standards have been adopted for nitrate (or other nutrients) with respect to this problem.

No violations of the un-ionized ammonia CMC criteria (neither salmonid nor nonsalmonid) were calculated for any of the 74 sites. We estimated compliance with the CCC criteria by comparing the instantaneous values computed for the CMC calculations with the 30-day CCC criteria given in the EPA report. Using this more conservative method of CCC calculation, most stations (60 of 74) still had no violations. Thirty-day average CCC violations may occur during winter months at certain stations. Examples include the Illinois River at Marseilles, IL (14% noncompliance); the Illinois River at Valley City, IL (3%); the Mississippi River at Nininger, MN (8%); and the Rock River at Joslin, IL (3%).

Minnesota River Basin

We used the Minnesota River as a case study to examine water quality violations in a highly agricultural area (see Figure 3.5) where more detailed seasonal data are available. Dissolved oxygen and nitrate data from eight tributary stations and seven mainstem stations from 1968 to 1994 were analyzed (Table 3.6, adapted from Mulla and Mallawatantri 1998). Data on pH and un-ionized ammonia were not available.

Violations of the DO standard occurred in three tributary stations, primarily during the winter: the Pomme de Terre River had 15% noncompliance; the Watonwan River, 20%; and the Blue Earth River at Blue Earth, 13% during the winter months. The Pomme de Terre River also had DO violations during July and August (22% noncompliance). DO violations on the Minnesota River mainstem were rare at upstream stations, but there was 2% noncompliance at Shakopee and 4% at Fort Snelling (near the mouth of the river). Figure 3.20 illustrates the relationship between measured DO and water temperature on the Minnesota River at Jordan. The data show a clear trend of decreasing concentrations with increasing temperature, as expected from the solubility–temperature relationship for dissolved oxygen. The wide scatter in the data ($r^2 = 0.52$), especially at low temperatures, indicates that factors other than temperature have important influences on DO concentrations. The lowest DO values (~5 mg/L), for example, occurred during the winter at water temperatures just above freezing and may have been caused by an ice cover that prevented atmospheric reaeration. The highest DO concentration (22 mg/L), which also occurred during cold weather (February 1990) is associated with the highest chlorophyll *a* concentration (454 µg/L) recorded at this site (cf. Figure 3.21).

Violations of the nitrate drinking-water standard were generally low for tributaries in the western part of the basin, but were numerous for Watonwan River and Blue Earth River stations. Violations at these stations occurred primarily during the spring and summer months. Peak violations occurred during May and June: 33% noncompliance for the Watonwan River, 66% for the Blue Earth River at Blue Earth, and 52% for the Blue Earth River at Mankato. Nitrate violations were minimal on the Minnesota River mainstem for stations above Mankato, but a significant number of violations were found below Mankato: 10% at St. Peter, 11% at Jordan (compared with 8% in the USGS database), and 5% at Fort Snelling. The reduction in noncompliance between Jordan and Fort Snelling may reflect dilution by lower nitrate waters from the suburban Twin Cities or assimilation of nitrate by phytoplankton in the river, which is fairly broad and slow-moving in the stretch between these two sites.

TABLE 3.6. Violations of water quality standards over 1968–94 for nutrient-related variables for Minnesota River Basin sampling stations.[1]

Station	Parameter	Percent Violation of Water Quality Standards by Time Period					
		Mar-Apr	May-Jun	Jul-Aug	Sep-Oct	Nov-Feb	Annual
Pomme de Terre River	NO_3+NO_2	0	0	0	0	0	0
(Appleton)	DO	7	4	22	5	15	11
Chippewa River	NO_3+NO_2	------------------------------Limited Data--------------------------					
(Montevideo)	DO	0	0	0	0	0	0
Yellow Medicine River	NO_3+NO_2	0	0	0	0	0	0
(Granite Falls)	DO	3	0	0	0	0	1
Redwood River	NO_3+NO_2	4	3	3	10	0	4
(Redwood Falls)	DO	0	3	0	0	0	1
Cottonwood River	NO_3+NO_2	7	7	0	0	0	3
(New Ulm)	DO	0	0	0	0	0	0
Watonwan River	NO_3+NO_2	22	33	16	5	8	17
(Garden City)	DO	0	0	0	0	20	5
Blue Earth River	NO_3+NO_2	38	66	38	12	10	31
(Blue Earth)	DO	5	0	0	0	13	4
Blue Earth River	NO_3+NO_2	29	52	13	11	13	23
(Mankato)	DO	0	0	0	0	0	0
Minnesota River	NO_3+NO_2						0
(Milan)	DO						4
Minnesota River	NO_3+NO_2						0
(Morton)	DO						0
Minnesota River	NO_3+NO_2						1
(Courtland)	DO						1
Minnesota River	NO_3+NO_2						10
(St. Peter)	DO						1
Minnesota River	NO_3+NO_2						11
(Jordan)	DO						0
Minnesota River	NO_3+NO_2						na
(Shakopee)	DO						2
Minnesota River	NO_3+NO_2						5
(Ft. Snelling)	DO						4

[1]*See text for values of water quality standards.*

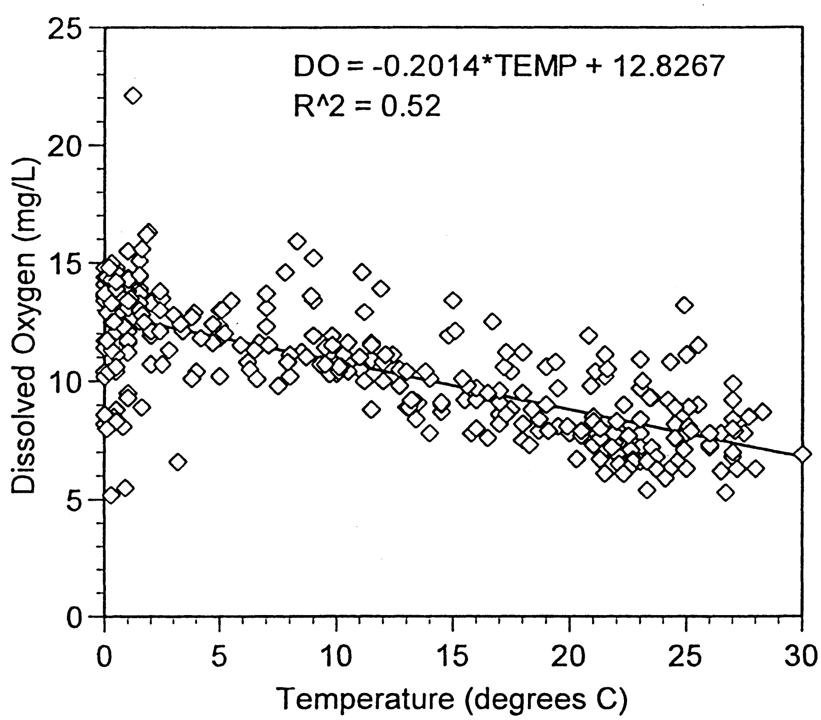


FIGURE 3.20. Dissolved oxygen concentrations versus water temperature in the Minnesota River at Jordan for 1980–92. NOTE: The dissolved oxygen saturation line is approximately equal to the shown regression line (DO saturation = 14.6 mg/L at 0°C, = 7.6 mg/L at 30°C). *(Data from Metropolitan Council Environmental Services, St. Paul, MN.)*

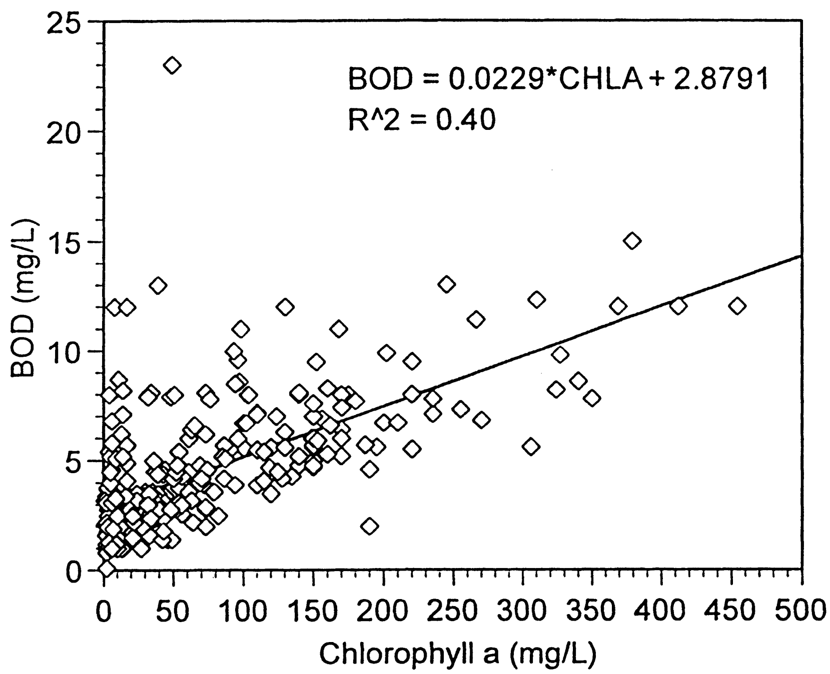


FIGURE 3.21. Five-day biochemical oxygen demand (BOD_5) versus chlorophyll *a* concentrations in the Minnesota River at Jordan for 1980–92. NOTE: The highest measured BOD (23 mg/L) is from October 1989, during a drought with one of the lowest recorded discharge values (230 cfs). *(Data from Metropolitan Council Environmental Services, St. Paul, MN.)*

State 305(b) reports

We examined the 1996 305(b) reports for Illinois (IEPA 1996), Indiana (IDEM 1996), Iowa (IDNR 1997), Minnesota (MPCA 1994), Missouri (MDNR 1996), Ohio (OEPA 1996), and Wisconsin (WDNR 1996), as well as the U.S. EPA 1996 305(b) Report to Congress (USEPA 1998a). The categories listed in these reports differ from those given in the sections above on the Mississippi and Minnesota River basins (Table 3.7). Furthermore, it is evident from the state 305(b) reports that different assessment techniques and levels of completeness make interstate comparisons invalid. For example, the Ohio data are a composite of data from 1988–96, data from some of the other states are for 1994–95, and the Minnesota report does not cover all of the river basins in the state. Thus, the data in these reports must be used with caution. We report the summary results in Table 3.7, with no further effort to make interstate comparisons.

TABLE 3.7. Nutrient-related water quality impairment of rivers and streams for selected Mississippi River Basin states from 305b reports for 1996.

State	Percent of Monitored Rivers and Streams Fully Supporting Resource Uses			Total Impaired Miles for Assessed Rivers and Streams					
	Aquatic Life Support	***Fish Consumption***	***Swimmable***	***Assessed Miles***	**Unionized Ammonia**	***pH***	***Organic Enrichment/DO***	***Nutrients***	***Agricultural Land Use***
AR	62	96	80						
CO	--------------80 Total--------------								
IL	54	81	30	28,454	466 (1.6)	316 (1.1)	2,446 (8.6)	11,760 (41)	11,361 (40)
IN	77	0	18	8,355	145 (1.7)	19 (0.2)	799 (9.6)	91 (1.1)	740 (8.9)
IA	0	71	2	10,140	61 (0.6)	na	47 (0.5)	47 (0.5)	3,349 (33)
KS	91	72	0						
KY	73	92	14						
LA	71	na	63						
MN	10	na	30	3,441	253 (7.4)	465 (13.5)	324 (9.4)	2,139 (62)	468 (14)
MI	2	27	9						
MO	51	99	100	21,015	24 (0.1)	27 (0.1)	61 (0.3)	1 (0)	7,111 (34)
MT	19	na	71						
NB	25	33	22						
ND	9	0	16						
OH	39	na	57	6,560	240 (3.7)	198 (3.0)	1,368 (6.5)	536 (8.2)	1,471 (22)
OK	9	na	20						
PA	82	82	82						
SD	17	na	43						
TN	73	99	92						
TX	91	98	73						

WV	38	24	72						
WI	33	77	na	19,898	255 (1.3)	5 (0.03)	1,826 (9.2)	3,060 (15.4)	4,431 (22)
WY	37	97	90						

The Wisconsin report (WDNR 1996) specifically warns that "…[t]he numbers generated by the U.S. EPA Waterbody System to answer questions the data were not gathered to address are not necessarily indicative of the condition of Wisconsin water quality. We are very concerned that these tallies may be misinterpreted and misunderstood by members of the public, and misapplied in a nationwide database." The same caveat applies to results from EPA's national 305(b) report (Table 3.7)—i.e., different assessment protocols makes interstate comparisons invalid.

Because of these concerns, the task group eschewed making quantitative comparisons among the states. Nonetheless, results from the state reports indicate that most states report only small fractions of their river miles as suffering from impaired use because of pH or un-ionized ammonia violations. However, in several MRB states, roughly 10% of the river miles are reported to be impaired because of organic enrichment/dissolved oxygen problems, and a wide range of use impairment (0% to > 60%) is reported because of nutrient problems. The *1996 305(b) Report to Congress* (USEPA 1998a) indicates that the percentages of monitored river miles fully supporting three important resource uses (aquatic life support, fish consumption, and swimming) vary widely among the states of the MRB. Again, differences in how states assess and report this information in their 305(b) reports make interstate comparisons impossible. Nonetheless, the summary results indicate that substantial percentages of the assessed river miles in many states did not fully support one or more of these important uses in the near past.

Summary

Overall, the above analysis indicates that rivers in the MRB generally meet ambient water quality standards for substances affected by nutrient loadings and concentrations (i.e., dissolved oxygen, pH, nitrate, and un-ionized ammonia). On this basis, it is reasonable to conclude that reductions in nutrient loadings to the rivers would not have a significant effect on the extent of compliance with the standards for these water quality variables. Several cautions need to be noted, however, that limit the strength of this conclusion:

- First, with regard to dissolved oxygen, substantial daily variations in concentration can result from diel changes in primary production and respiration during a 24-hour (day–night period). Minimum DO concentrations generally occur at or just before dawn, and it is unlikely that the monitoring data used in our analysis reflect measurements at that time of day.
- Second, the USGS and 305(b) databases are heavily skewed toward medium and large rivers. Therefore, the above conclusion may not apply to MRB streams and small rivers. Indeed, long-term monitoring data for several branches of the Greater Blue Earth River, a tributary of the Minnesota River, show frequent violations of the nitrate (drinking water) standard, especially in spring and summer (Aplikowski 1999), and such violations are likely for similar sized streams and small rivers in other parts of the Corn Belt.
- Third, the 305(b) reports give aggregate information for each state and do not separate the miles of impaired use by river basin. Only portions of some of the states in Table 3.7 lie within the MRB.
- Fourth, the number of sites for which long-term water quality information is available is small compared with the total lengths of rivers in the MRB, and the sampling sites were not established on the basis of a random or "probability-based" sampling design. Consequently, results from the sampling sites cannot be extrapolated to the system as a whole with known statistical reliability.
- Finally, and probably most important, numerical water quality standards have not yet been developed for nutrients in surface waters. The only available standards for nitrogen forms are related to toxicity to aquatic organisms (un-ionized ammonia) and humans (nitrate). The human health-based standard for nitrate is much higher than the concentrations associated with water quality problems induced by eutrophication. Consequently, the low frequency of noncompliance data does not necessarily mean that water quality conditions in the rivers are satisfactory (see Section 3.2.6.3).

3.2.6.2 REDUCTIONS IN CBOD AND NBOD

Insofar as violations of DO standards appear to be infrequent events in MRB waters, levels of oxygen-demanding, biodegradable organic matter in the rivers do not appear to be a major cause for concern. Nonetheless, nutrient-source reductions in the basin should lead to somewhat lower levels of biochemical oxygen demand (BOD) in the rivers for several reasons. First, some declines in inputs of biodegradable organic matter are likely as a result of improvements in point-source waste treatment, improved management and treatment of urban stormwater, and changes in agricultural practices to decrease N and P export. Second, lower concentrations of nutrients in the rivers should cause a reduction in primary production, leading to lower levels of autochthonous dissolved and particulate organic carbon in the water column (thus decreasing the carbonaceous BOD or CBOD). Third, lower nutrient input rates will tend to reduce ammonium concentrations in the rivers, which will decrease the oxygen demand caused by nitrification of ammonium to nitrate (so-called NBOD).

A review of available databases on the quality of MRB waters revealed an absence of data on BOD values in the rivers. The exception was at locations near discharges from municipal wastewater treatment plants, where “compliance” monitoring is required by pollution control agencies. Such data are not representative of BOD levels in the river system as a whole, and we are unable to estimate typical CBOD concentrations in MRB rivers and streams.

The potential role of primary production within the rivers in generating biodegradable organic matter (i.e., CBOD) was evaluated by plotting a large data set on chlorophyll *a* concentrations and CBOD values for the Jordan site on the Minnesota River (Figure 3.21). Although a positive relationship was found, the regression explained only about 40% of the variance, and factors other than the abundance of algae (as represented by chlorophyll *a*) clearly are affecting the concentration of CBOD at this site. The highest CBOD reported at this site (23 mg/L) occurred in October 1989 during a period of very low flow (230 cfs, which is < 5% of the average flow for the period 1980–92).

Even at chlorophyll concentrations of several hundred µg/L, CBOD tends to be < 14 mg/L. It is interesting to note that the highest observed chlorophyll *a* concentration at the site (a very high value of 454 µg/L) was associated with a BOD of 12 mg/L, and also with the highest DO measured at the site (22 mg/L; see Figure 3.20). The trends shown in Figures 3.20 and 3.21 suggest that high concentrations of chlorophyll *a* found in MRB rivers under present conditions generally do not produce CBOD levels high enough to induce DO problems in the rivers.

Routine monitoring data are available for ammonium concentrations at many sites in the rivers, and it is a simple matter to calculate the potential NBOD that these concentrations represent. From simple stoichiometric relationships one can show that 4.5 mg of O_2 are consumed per mg of NH_4^+-N oxidized to nitrate-N. Ammonium concentrations in MRB waters typically are low, except downstream from point-source discharges. For example, ammonium concentrations in the Mississippi River at Nininger, MN, which is downstream from the effluent discharge for the main Minneapolis–St. Paul wastewater treatment plant, are the highest of all 74 stations in the USGS database for the MRB. The following statistics apply ammonium concentrations at this site for the period October 1977–November 1992: minimum = 0.02; mean = 0.48; maximum = 1.70 (all values in mg N/L). On this basis, the potential NBOD at this site over this 15-year period ranged from < 0.1 mg/L to 7.7 mg/L (mean = 2.2 mg/L).

Analogous figures for the Mississippi River at Alton, IL, just below the confluence of the Missouri and Mississippi Rivers, are:

- NH_4^+-N: minimum = 0, mean = 0.17, maximum = 1.4 mg N/L; and
- NBOD: minimum = 0, mean = 0.8, maximum = 5.3 mg/L.

These figures are fairly typical for sites in the 74-station database. Many sites have mean concentrations below 0.1 mg N/L, and only a few have concentrations above 0.2 mg N/L. These values should not cause oxygen depletion by NBOD within the river.

3.2.6.3 REDUCTION IN EXCEEDANCES OF NUTRIENT-BASED, TROPHIC-STATE CRITERIA

Although legally binding numeric standards have not been established in any state for nutrients in flowing water systems or lakes, efforts toward developing such standards have been ongoing for several years. At present, many states have a non-numeric standard that, in essence, says nutrients must not be added to a water body to the extent that they cause an imbalance in its natural flora and fauna. Science-based numerical criteria for nutrient concentrations have long been available, however, to classify lake ecosystems according to trophic state, and a substantial literature is available that relates these criteria to the likelihood of specific water quality problems, such as frequency of nuisance algal blooms and poor water clarity. The water quality problems that can result from river eutrophication are summarized in Table 3.8.

In contrast to lakes, few classification methods exist in the literature to evaluate the trophic status of stream ecosystems in quantitative terms. Dodds et al. (1998) recently proposed a simple classification scheme for streams that includes concentration boundaries approximately defining mesotrophic and eutrophic conditions. This classification scheme is based on measurements of chlorophyll, total P, and total N from nearly 1,400 rivers and streams in North America, Europe, and New Zealand. The rivers and streams drain land-cover and land-use conditions ranging from pristine forests to intensive agriculture. Concentrations of 1.5 mg/L for total N and 0.075 mg/L for total P were proposed as approximate boundaries separating mesotrophic and eutrophic conditions in these flowing waters (Dodds et al. 1998).

TABLE 3.8. Potential effects of nutrient enrichment on water quality in rivers and streams.

Suspended Algae	**Periphyton**
Increased biomass and changes in species composition of suspended algae. Taste and odor problems. Blockage of intake screens and filters. Disruption of flocculation and chlorination processes at water treatment plants. Reduced water clarity. Impairment of recreational use. Fish kills. Harmful diel fluctuations in pH and dissolved oxygen concentrations.	Increased biomass and changes in species composition. Blockage of intake screens and filters. Floating mats of algae. Restriction of swimming and other water-based recreation. Slippery surfaces making wading or standing dangerous. Fouling of submerged lines and nets.

To evaluate the potential in-stream benefits of reductions in nutrient inputs to rivers and streams in the Mississippi River Basin, we applied the concentration boundaries proposed in this simple classification method. We analyzed stream concentrations of total nitrogen (TN) and total phosphorus (TP)] in the interior watersheds of the MRB (R.A. Smith et al. 1997) in relation to estimated exceedance rates of the proposed stream trophic criteria for nutrients (Dodds et al. 1998). The resulting expressions provide estimates of the expected response of stream nutrient exceedances to hypothetical reductions in stream concentrations that would result from reductions in nutrient loads from watershed sources. We did not evaluate the nature of the relations between stream nutrient concentrations and sources. Our analysis here assumes that the hypothetical percentage reductions in stream nutrient concentrations would result from a similar percentage reduction in source loads if applied equally to all watershed sources.

The analysis presented here is based on estimates of mean stream concentrations of TP and TN for 2,050 hydrologic cataloging units (HCUs) in the United States (Seaber et al. 1987) obtained from a statistical investigation of long-term monitoring data using the SPARROW water-quality model (R.A. Smith et al. 1997). HCUs are systematically defined drainages for the United States and are generally representative of larger river systems (Seaber et al. 1987). The HCUs are sub-watersheds located within the 18 water resource regions of the conterminous United States (Figure 3.22). The analysis included 838 HCUs, averaging approximately 3,600 km^2, in the six water resource regions of the MRB—Ohio, Tennessee, Missouri, Arkansas–Red, and Upper and Lower Mississippi.

Figures 3.23 and 3.24 describe the relationships between the median nutrient concentration in each water resource region and the percentage of HCUs in each region that exceeds the mesotrophic/eutrophic boundary proposed for TN (Dodds et al. 1998). We selected a criterion concentration of 0.10 mg/L for TP, somewhat greater than that proposed by Dodds et al. (1998), because of the availability of these estimates from a previous analysis (R.A. Smith et al. 1997).

There is significant spatial variation in the stream-water concentrations of total N and total P across the six water resource regions studied here. Concentrations of TP and TN are lowest in the Tennessee, Ohio, and Lower Mississippi (LM) regions, where HCU concentrations commonly range from 0.06 to 0.35 mg/L for TP, and from about 0.19 to 2.6 mg/L for TN. Median concentrations in the latter three regions (Figures 3.23, and 3.24) range from 0.09 to 0.16 mg/L for TP and from 0.77 to 1.20 mg/L for TN. HCU concentrations in the Upper Mississippi, Arkansas–Red, and Missouri regions are significantly higher: about 0.20–0.80 mg/L for TP and 0.12–9.0 mg/L for TN. Median concentrations in the latter three regions are 0.40–0.50 mg/L for TP and 2.7–4.0 mg/L for TN (Figures 3.23 and 3.24).

There is also significant spatial variation in the degree to which HCUs in the six water resource regions of the MRB exceed the trophic-state criteria proposed by Dodds et al. (1998). For example, low-to-moderate proportions of the HCUs monitored in the Tennessee (29%), Ohio (49%), and Lower Mississippi (53%) exceed the eutrophic criterion for TP. Similarly, low-to-moderate proportions of the HCUs in these three regions exceed the proposed criterion for TN (Tennessee, 16%; OH, 40%; LM, 35%; see Figures 3.23 and 3.24). In contrast, much higher frequencies of exceedance are found in the Upper Mississippi (UM), Arkansas–Red, (AR), and Missouri regions, where about 80% of the HCUs exceed the TP trophic criterion, and 70–75% exceed the TN criterion.

FIGURE 3.22. Major water resource regions in the conterminous United States.

According to the regression fits in Figures 3.23 and 3.24, a nonlinear relationship exists between the frequency of exceedances of the stream trophic criteria and median nutrient concentrations in a water resource region. The slopes of the relations indicate that the response of the exceedance frequency to changes in nutrient concentrations is less than proportional (i.e., slope < 1). A greater response to a unit change in concentration is anticipated for cataloging units in the Ohio, Tennessee, and LM regions than for those in the UM, AR, and Missouri regions (see Table 3.9). In the former regions, where nutrient concentrations are generally lower, a 1% change in median concentration is expected to result in about a 0.4% change in the exceedance rate for TP and a 0.7% change for TN. These values are about twice the expected changes in exceedance frequencies for the UM and Missouri regions. A greater percentage reduction in concentrations thus is required in the UM, AR, and Missouri regions than in the Ohio, Tennessee, and LM to obtain a similar percentage reduction in the exceedance rates for TN and TP. For example, a 30% reduction in TP concentrations is required in the UM, AR, and Missouri regions to obtain a 10% reduction in the number of cataloging units that exceed the stream trophic criterion; whereas only a 15% reduction in TP concentration is required in the Ohio, Tennessee, and LM regions to achieve a 10% reduction in the rate of exceedance. In general, a slightly greater percentage reduction in TN concentrations is required than in TP concentrations to achieve a similar percentage reduction in exceedance of the stream trophic criteria.

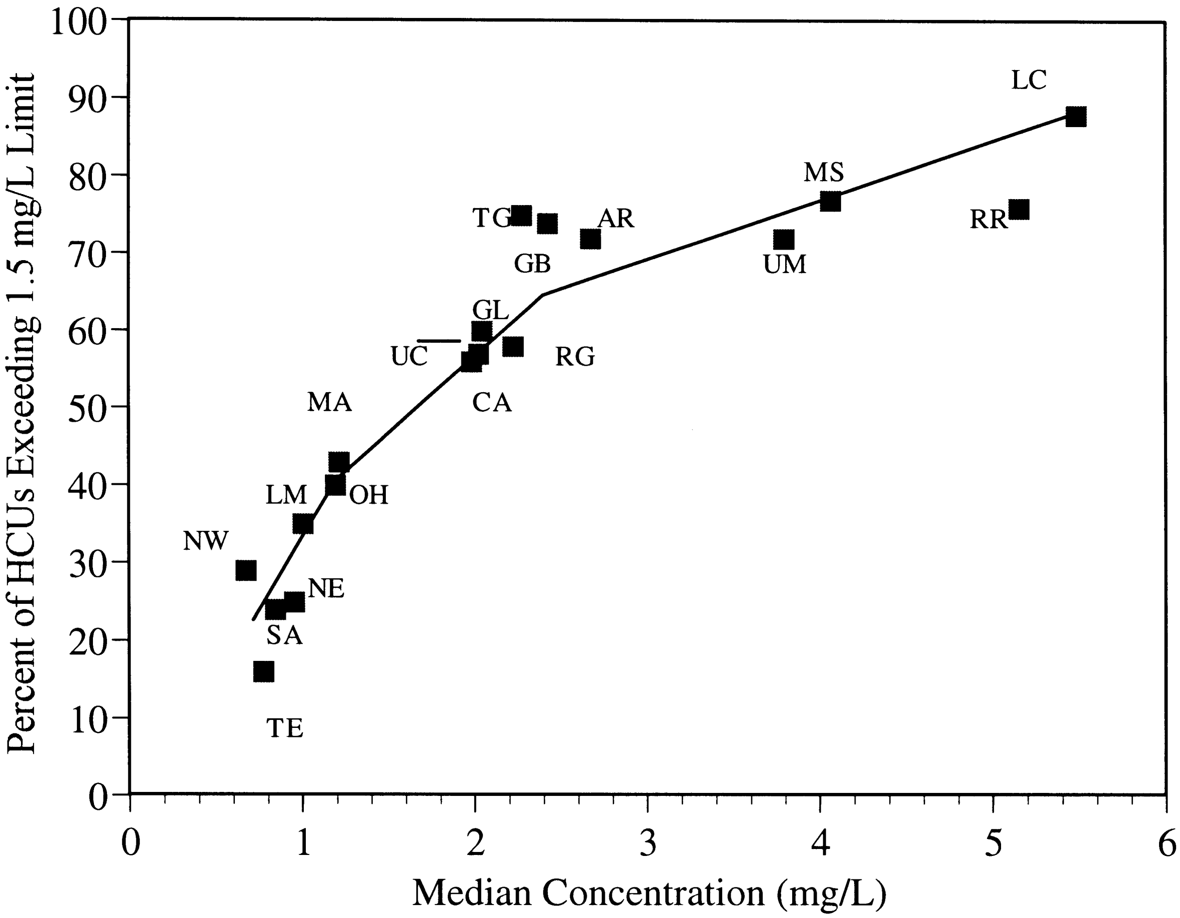


FIGURE 3.23. Percentage of cataloging units exceeding the stream eutrophic state criterion for TN versus the median TN concentration in the 18 water resource regions of the conterminous United States. NW = Pacific Northwest; TE = Tennessee; SA = South Atlantic Gulf; NE = New England; LM = Lower Mississippi; OH = Ohio; MA = Mid-Atlantic; CA = California; UC = Upper Colorado; GL = Great Lakes; RG = Rio Grande; TG = Texas–Gulf; GB = Great Basin; AR = Arkansas–White–Red; UM = Upper Mississippi; MS = Missouri; RR = Souris–Red–Rainy; and LC = Lower Colorado.

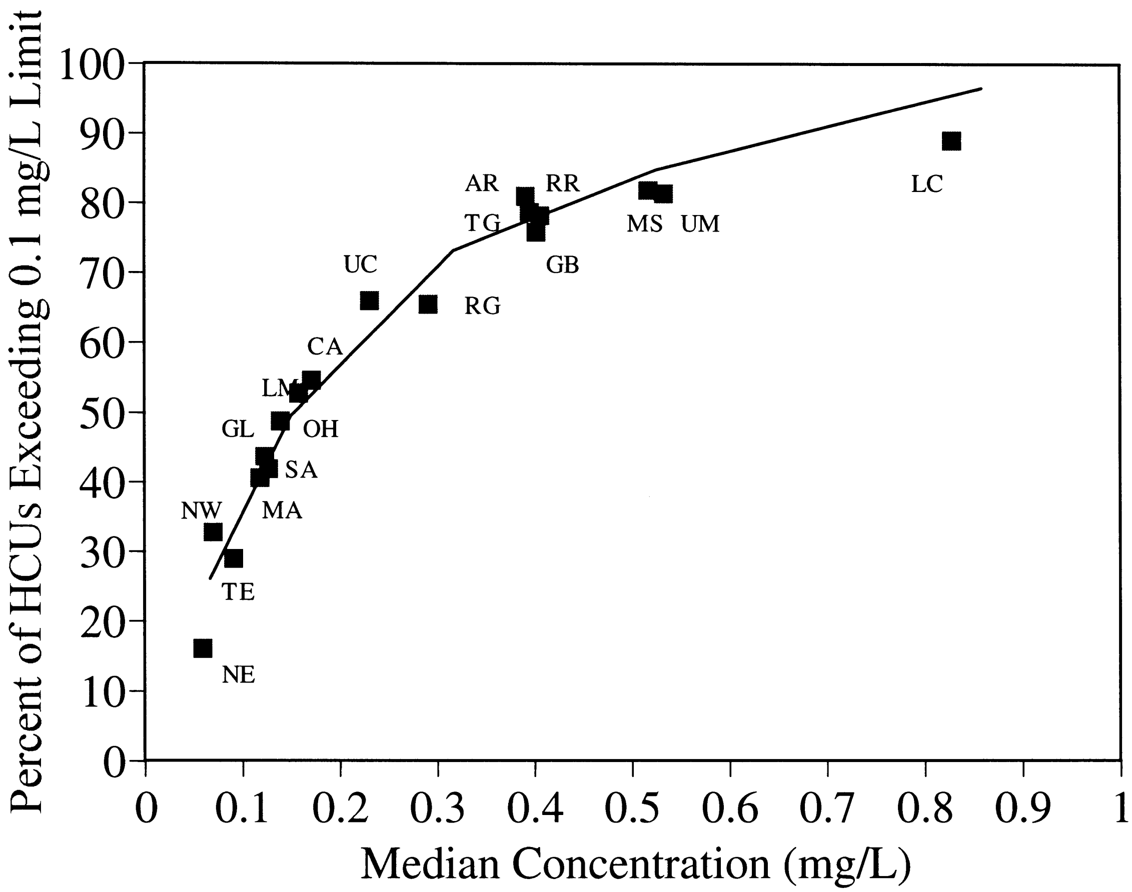


FIGURE 3.24. Percentage of cataloging units exceeding the stream eutrophic state criterion for TP versus the median TP concentration in the 18 water resource regions of the conterminous United States. NW = Pacific Northwest; TE = Tennessee; SA = South Atlantic Gulf; NE = New England; LM = Lower Mississippi; OH = Ohio; MA = Mid-Atlantic; CA = California; UC = Upper Colorado; GL = Great Lakes; RG = Rio Grande; TG = Texas–Gulf; GB = Great Basin; AR = Arkansas–White–Red; UM = Upper Mississippi; MS = Missouri; RR = Souris–Red–Rainy; and LC = Lower Colorado.

TABLE 3.9. Estimated change in rates of exceedance of stream trophic-state criteria for TP and TN in hydrologic cataloging units of six Mississippi River sub-basins as a function of change in stream concentrations.

Water Resource Regions	% Change in Exceedance Rate for a 1% Change in Concentration[1]		% Reduction in Concentration Required for a 10% Reduction in the Rate of Exceedance	
	TP	TN	TP	TN
Lower Mississippi, Tennessee, Ohio	0.66	0.43	15	20
Upper Mississippi, Missouri, Arkansas–Red	0.33	0.28	30	35

[1]Based on the slope of the exceedance and concentration relationships in Figures 3.23 and 3.24 for the indicated water resource regions.

The above analysis for rivers and streams in the MRB can be placed in broader perspective relative to trophic conditions in streams across the United States. The mean TP concentration of water reported by R.A. Smith et al. (1987) for 381 riverine sites in the continental United States is 130 µg/L, a value greatly exceeding the proposed mesotrophic–eutrophic boundary of 75 µg/L (Dodds et al. 1998). Similarly, R.A. Smith et al. (1993) concluded that 48% of 410 water quality monitoring stations failed to meet EPA's 1976 proposed criterion of 100 µg/L (USEPA 1976). In another study, R.A. Smith et al. (1997) found that 61% of 2,048 HCUs failed to meet this criterion. Collectively, this evidence suggests that nutrient conditions are relatively poor in a majority of U.S. streams and rivers in relation to the currently available trophic criteria for streams.

3.2.6.4 EFFECTS ON PLANKTON COMMUNITIES IN THE RIVERINE ECOSYSTEM

Nutrient limitation of algal growth in rivers and streams

Although most of the research on eutrophication during the past several decades has focused on lakes and reservoirs, nutrient overenrichment of flowing waters is also a matter of great concern. For many years, flowing waters were perceived as nutrient-saturated, and the prevailing opinion was that such factors as light limitation and short hydraulic residence times should restrict or prevent algal response to nutrient enrichment in flowing waters. However, evidence from a wide variety of geographical locations indicates that many streams and rivers respond strongly to anthropogenic inputs of N and P (V.H. Smith 1998).

Some of the earliest evidence for such a response in streams comes from Huntsman (1948), who fertilized an oligotrophic stream in Nova Scotia, Canada, by immersing bags of agricultural NPK (nitrogen, phosphorus, potassium) fertilizer. Sites downstream of the fertilizer additions quickly exhibited increased abundance of attached filamentous green algae and fish. Ten years later, Correll (1958) performed a more controlled nutrient-enrichment experiment in a small Michigan stream. The TP concentration of the stream water was increased from background levels of < 8 µg/L to 70 µg/L at the enrichment site, and elevated TP concentrations were observed up to 4 km downstream. Periphyton growth on submersed artificial substrates increased threefold relative to unfertilized conditions, in response to these nutrient additions (Correll 1998).

Many other more recent studies have confirmed the stimulatory effects of N and P inputs on algal growth in streams. For example, Stockner and Shortreed (1978) used streamside wooden troughs to examine the effects of nutrient availability on periphyton growth in a shaded British Columbia stream. One experimental trough was enriched for 52 days with P alone; another, with N alone; and a third, with N and P. Although little response occurred in the +N trough, P additions resulted in almost a five-fold increase in attached algal biomass, and additions of both N and P caused almost eight-fold increases in growth relative to the control trough.

Similar results were found by Elwood et al. (1981), who enriched two reaches of a shaded oligotrophic Tennessee stream with inorganic P for 95 days. Stream-water P concentrations increased by over an order of magnitude, and this resulted in significantly increased benthic algal biomass, higher rates of detritus decomposition, and greater abundance of benthic macroinvertebrate consumers. Peterson et al. (1983, 1985, 1993) observed strong increases in downstream periphyton biomass following the addition of inorganic P to an Alaskan tundra river, and the stimulation of primary productivity cascaded into higher consumer populations in the stream (Hershey et al. 1988). Krewer and Holm (1982) and Horner et al. (1990) also found that the growth of periphyton in laboratory stream channels was strongly dependent on the concentration of phosphorus in the flowing water.

Phosphorus is not the sole limiting nutrient in streams and rivers, however. As noted in several of the studies cited above, enrichment with both N and P often produces higher algal yields than additions of N or P alone, and these data suggest that both N and P can be co-limiting to algal communities in streams. In other streams, N alone may be the primary limiting nutrient. For example, both whole-stream N enrichment studies (Gregory 1980) and artificial trough experiments (Triska et al. 1983) suggested potential N limitation, but in both of these studies the responses of periphyton to nutrient enrichment were strongly damped by low light availability. Strong limitation of benthic algae by nitrogen also was inferred for streams in Arizona (Grimm and Fisher 1986), California (Hill and Knight 1988), Missouri (Lohman et al. 1991), and Montana (Lohman and Priscu 1992). Although studies on nutrient limitation in suspended riverine algae are much less common, Köhler and Gelbrecht (1998) recently reported evidence for N and P limitation of suspended algal growth in a large river system in Germany.

The studies above suggest that rather than being rare, nutrient limitation of algal growth in flowing waters is common and widespread. This hypothesis is supported by numerous comparative statistical analyses, which confirm that nutrient enrichment of streams and rivers typically is accompanied by increased biomass of suspended and/or benthic algae (Basu and Pick 1996; Dodds et al. 1997, 1998; Smart et al. 1985; Lohman et al. 1992; McGarrigle 1993; Soballe and Kimmel 1987; Van Nieuwenhuyse and Jones 1996; Welch et al. 1992). The production of algae per unit TP often is significantly lower in rivers than in lakes and reservoirs, however. This difference may be caused in part by the higher washout (loss) rate that flowing water may impose on suspended algal biomass (Soballe and Kimmel 1987; Van Nieuwenhuyse and Jones 1996), but lower availability of light (because of shading and higher levels of inorganic suspended solids than are typical of lakes) also may play a role.

Empirical support for the presence of washout effects on riverine algal biomass yields comes from the studies of Van Nieuwenhuyse and Jones (1996) and Lohman and Jones (1998), who found that differences among streams in their response to TN and TP could be accounted for in part by including watershed area as an additional explanatory variable (see equation 3–1 under the next subsection). The observed effects of area in these two studies were attributed to higher hydraulic flushing rates (short water-residence times) in lower-order streams. However, the data in Figure 3.25 suggest that the effects of water-residence time on suspended algal biomass are not always strongly apparent in rivers. Except for three sites (including two with exceptionally short residence times of about one day), a highly significant positive relationship was evident between phytoplankton biomass and stream-water TP concentrations in the eutrophic River Bure, UK (Moss et al. 1989). Similarly, Basu and Pick (1996) did not find a significant effect of water residence time on suspended chlorophyll in streams that were fifth order and larger.

A parallel dependence of phytoplankton biomass on TP also has been observed in other rapidly flushed systems. For example, Hoyer and Jones (1983) observed a strong chl *a*–TP relationship for 96 reservoirs in Missouri and Iowa that encompassed a wide range of water-residence times (~8 days to 33 years). They found that high concentrations of inorganic suspended solids reduced the algal yields in these turbid reservoirs, but they did not observe a significant effect of flushing rate. Similarly, Soballe et al. (1992) observed a strong correlation between chl *a* and TP in 45 mainstem and tributary reservoirs operated by the Tennessee Valley Authority and the U.S. Army Corps of Engineers. The chl *a* produced per unit TP tended to be lower in the mainstem systems, but it is difficult to determine whether this difference resulted from higher concentrations of inorganic suspended solids, from shorter water-residence times in the mainstem reservoirs, or from a combination of both factors.

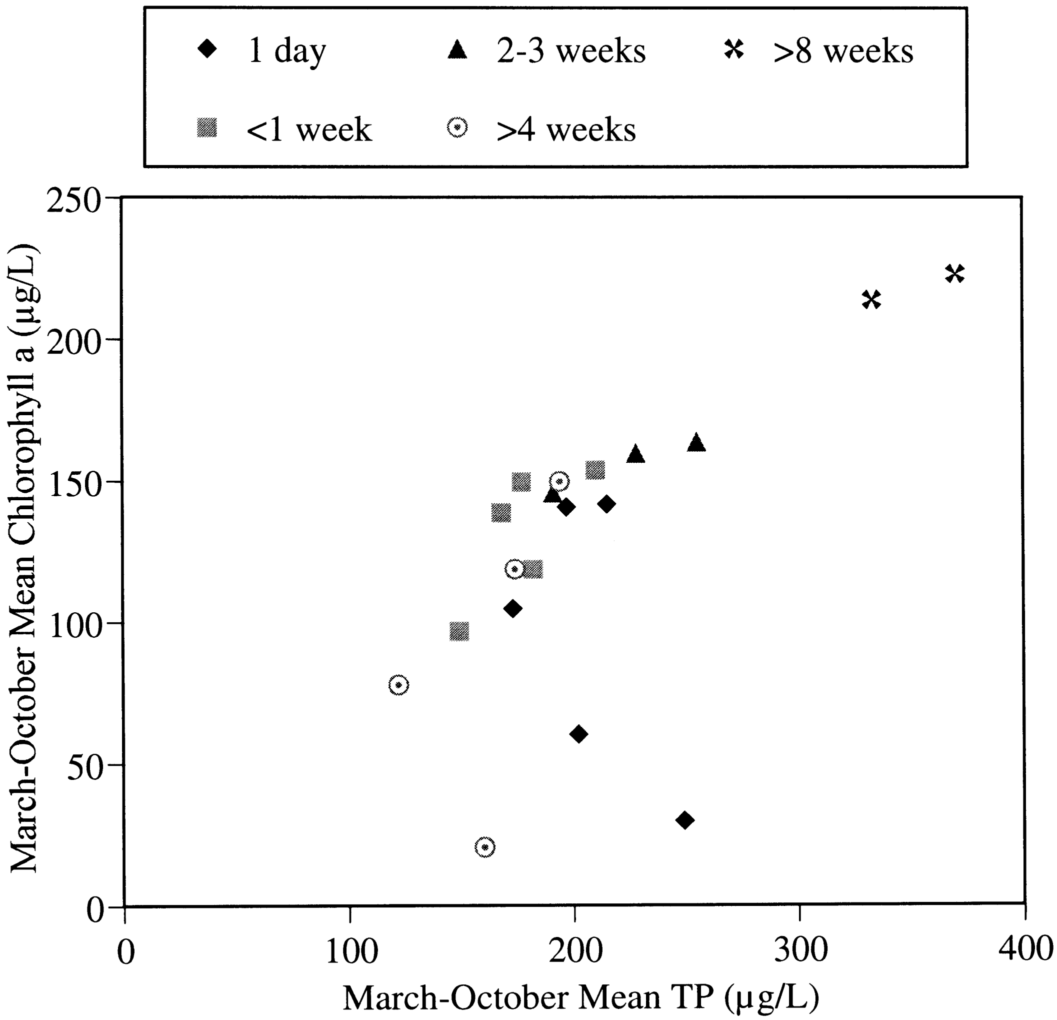


FIGURE 3.25. Mean chlorophyll *a* concentrations versus mean total P concentrations in the River Bure, UK, with sites classified by water residence time. *(Data replotted from Moss 1989.)*

Potential for nutrient limitation of phytoplankton in the MRB system

The panel was unable to locate any published or unpublished data on algal nutrient bioassays conducted on flowing waters in the MRB. As a result, we empirically assessed the degree to which N and P limit phytoplankton biomass in the MRB system, using a long-term data set on water quality for seven river sites in the Minneapolis–St. Paul area of the Upper MRB (provided by C. Larson, Metropolitan Council Environmental Services (MCES), St. Paul, MN), and a USGS data set for 837 cataloging units in the MRB. The methods used in the analyses here are based on the empirical approach used successfully in the management of eutrophication in lakes and reservoirs worldwide (OECD 1982; Reckhow and Chapra 1983; V.H. Smith 1998).

The extent of potential N and P limitation in the MCES data set was evaluated using the nutrient ratio guidelines proposed by Sakamoto (1966) for phytoplankton and later adopted by the OECD (1982):

Potentially Limiting nutrient	*TN:TP Mass Ratio in the Water*
N	< 10
N and P	< 10
P	> 17

The MCES data were screened by sampling date, and data from all May–September samples (inclusive) were sorted by their TN:TP ratio. For 4,289 total samples, the results were as follows:

Potentially Limiting Nutrient	*TN:TP Mass Ratio*	*Number of Samples*
N	< 10	1,003
N and P	10–17	1,456
P	> 17	1,830

Less than 25% of the samples (1,003/4,289 = 23.4%) exhibited potential limitation by N alone based on these criteria, and limitation by P alone was present in 43% of the samples. Limitation by both N and P was assumed to be present in 34% of the samples. Phosphorus thus appears to be the dominant limiting nutrient in this region of the MRB.

Estimates for mean annual TN and TP concentrations within the 837 cataloging units in the USGS data set were calculated from the SPARROW model and analyzed similarly. The results (Table 3.10) suggest that for the MRB as a whole, a majority of the waters fall into the combined N + P and P-limited class (percentage of waters with TN:TP > 10 = 69%), and 31% of the sites exhibited potential N limitation. Geographical differences occur in the distribution of potential nutrient limitation, however. The Upper Mississippi region had similar numbers of N- and P-limited waters (12% and 13%, respectively), and so did the Lower Mississippi region (28% and 26%, respectively). In contrast, waters in the Tennessee, Missouri, and Arkansas–Red regions appeared to be distinctly more N-limited, and sites in the Ohio River region had a somewhat greater percentage of P-limited than N-limited waters.

TABLE 3.10. SPARROW estimates of TN:TP ratios by major hydrologic region in the Mississippi River Basin.

Hydrologic Region	**No. of HCUs[1]**	**TN:TP Ratio**					**Percent of HCUs by Class**			
		Median	***25th[2]***	***75th[2]***	***Min.***	***Max.***	***<10***	***10-17***	***>17***	***sum >10***
Ohio	120	12.4	10.5	17.2	5.9	19.2	19.2	55.0	25.8	80.8
Tennessee	32	11.6	10.0	14.2	3.7	25.0	25.0	68.8	6.3	75.1
Upper Miss.	130	11.8	10.7	14.1	5.5	12.3	12.3	74.6	13.1	87.7
Lower Miss.	82	12.9	9.1	17.3	2.6	28.1	28.1	46.3	25.6	71.9
Missouri	302	10.8	9.5	12.8	5.4	36.1	36.1	60.3	3.6	63.9
Arkansas/Red	17	10.3	8.9	11.7	5.2	46.8	46.8	52.1	1.2	53.3
TOTAL MRB	**837**	**11.2**	**9.7**	**13.5**	**2.6**	**30.9**	**30.9**	**59.0**	**10.0**	**69.0**

[1]Hydrologic cataloging units. (See Seaber et al. 1987 for a description of HCUs.)
[2]Percentile of HCUs in the region.
Source: R.A. Smith et al. 1997.

The nutrient dependence of phytoplankton biomass in the MRB was also evaluated using the MCES data set. Mean values of TN, TP, chl *a*, and other water quality parameters were generated for the period May–September, and the data were analyzed graphically. As shown in Figure 3.26, data from six of the seven sites exhibited a positive relationship between chl *a* and TP concentrations.

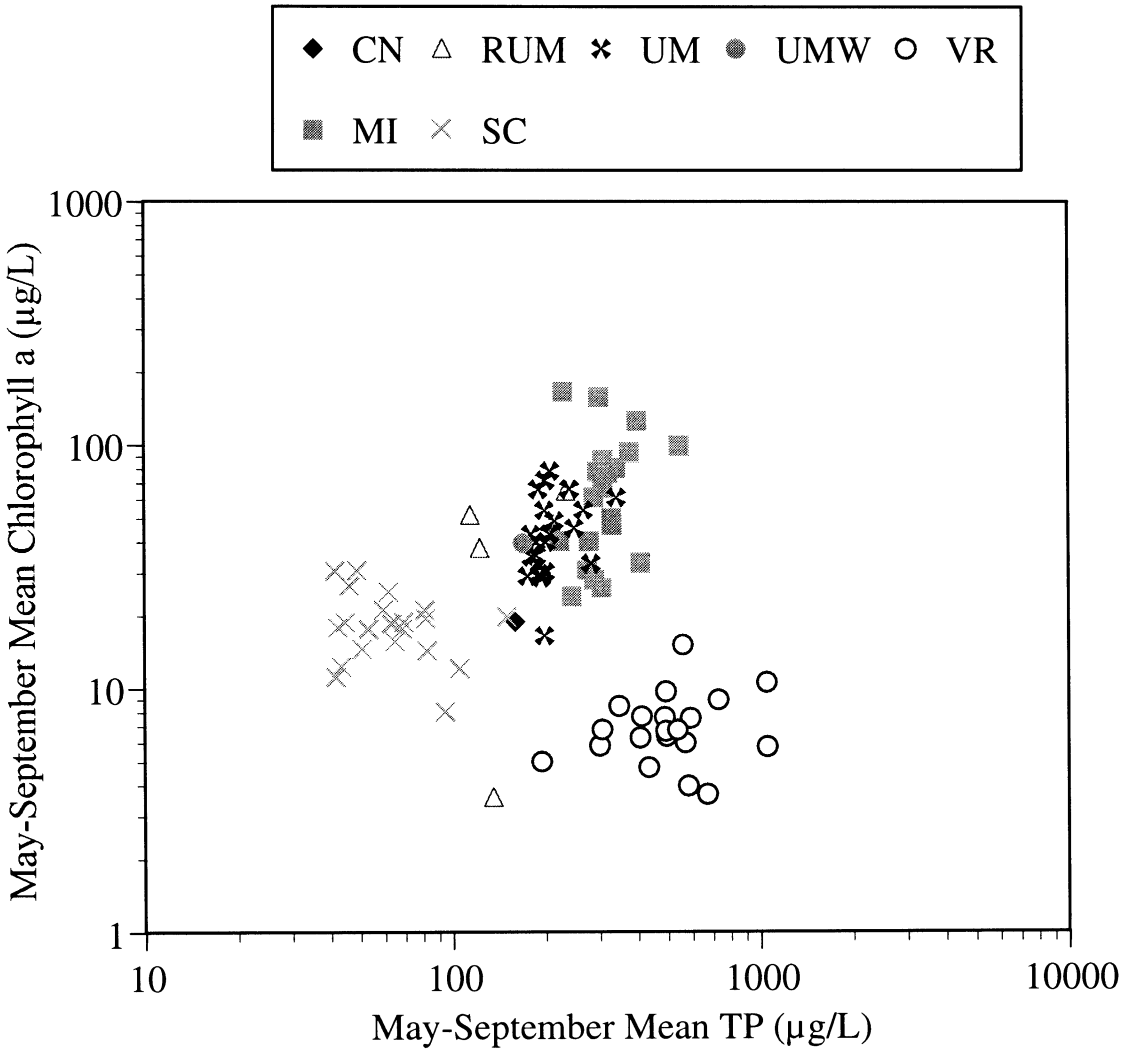


FIGURE 3.26. Mean chlorophyll *a* concentration for May–September versus mean total P concentration for the same period at sampling sites in the metropolitan Twin Cities area. *(Data from C. Larson, Metropolitan Council Environmental Services, St. Paul, MN, 1998.)* CN = Cannon River; MI = Minnesota River; RUM = Rum River; SC = St. Croix River; UM = Upper Mississippi River; UMW = Upper Mississippi River, Wisconsin channel (downstream of Hastings, MN); VR = Vermillion River.

Only the Vermillion River (VR) site deviated consistently from this pattern. The deviation does not appear to be due to N limitation, because no general tendency for the Vermillion River to exhibit low TN:TP ratios is evident (Figure 3.27).

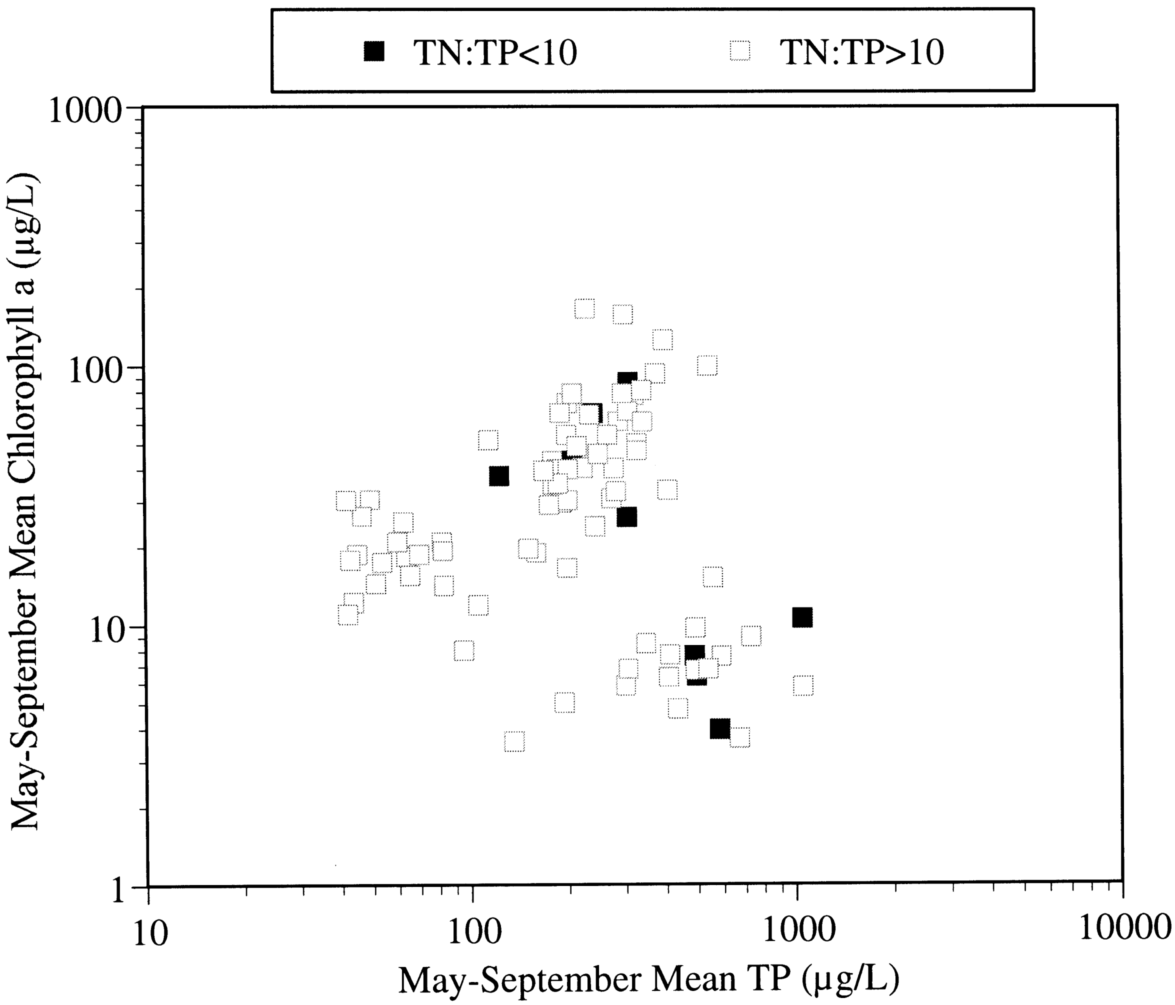


FIGURE 3.27. Replot of Figure 3.26 with data coded according to the ratio TN:TP.

Similarly, Figure 3.28 provides no evidence that the deviation reflects light limitation caused by high concentrations of inorganic suspended solids in the water. To obtain Figure 3.28, we sorted the data according to Hoyer and Jones' (1983) empirical criterion based on the ratio nonvolatile suspended solids:total phosphorus (NVSS:TP), which they developed for turbid reservoirs. If high concentrations of inorganic turbidity had caused the Vermillion River sites to deviate from the general trend, a clustering of potentially light-limited points (NVSS:TP > 0.13) should have been evident among the VR data, but this pattern is not observed. The Vermillion River on average is faster flowing and perhaps more consistently canopy-shaded than the six other UMR sites (C. Larson, personal communication). The observed deviations seen in Figure 3.26, thus, may be due to these two factors or to other variables not measured by the MCES that could not be included in this analysis. Further studies are needed to clarify these potentially confounding factors and quantify their effects on the overall relationship between TP and suspended algal biomass in the MRB.

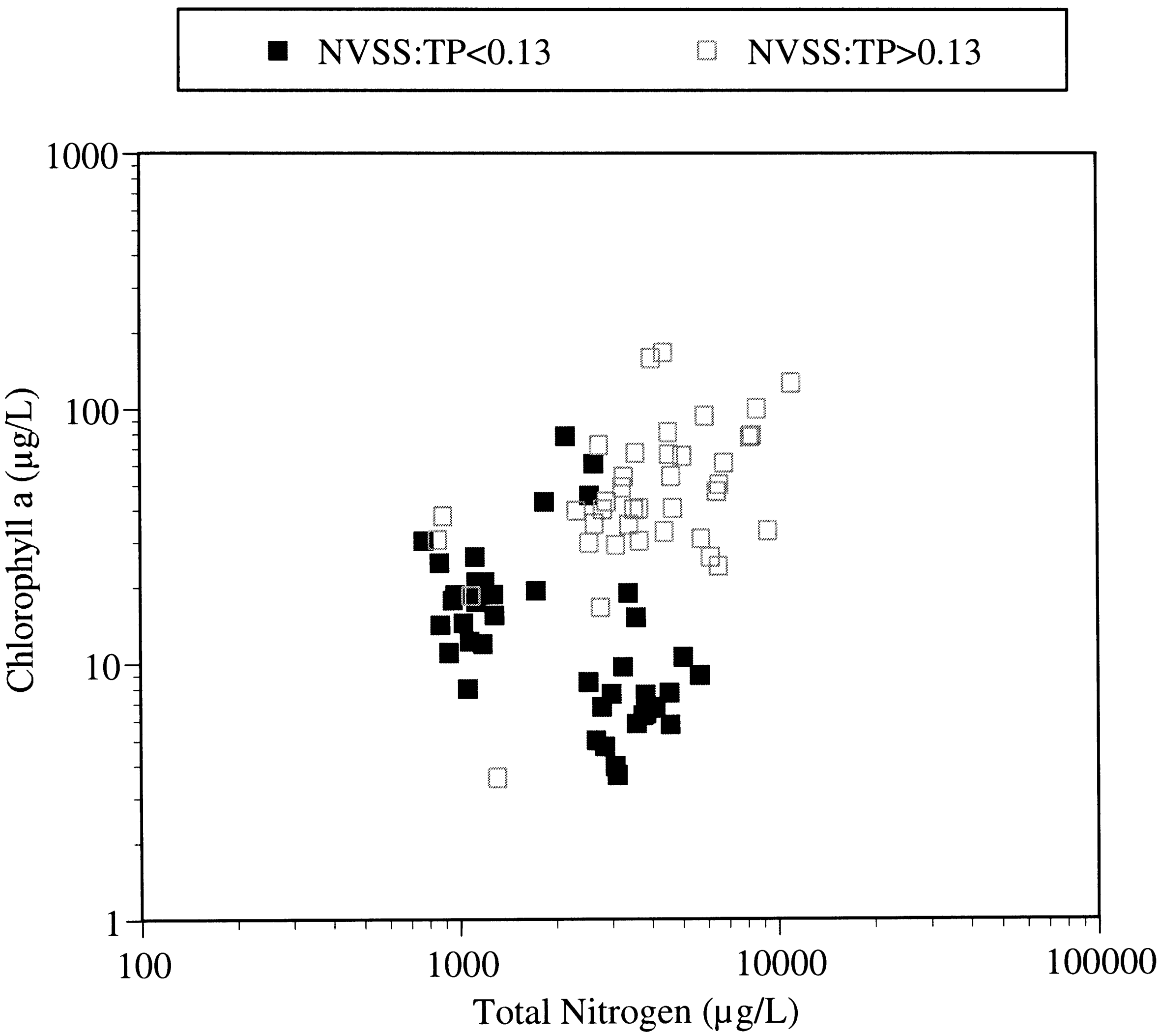


FIGURE 3.28. Replot of Figure 3.26 with data coded according to the ratio NVSS:TP.

The degree to which behavior of the UMR sites is consistent with algal responses in other systems was assessed by comparing the MCES data with two other databases: (1) the river database of Van Nieuwenhuyse and Jones (1996) (Figure 3.29) and (2) the turbid reservoir data set of Jones and Knowlton (1993) (Figure 3.30). Both comparisons show a strong consistency of the trends for the UMR sites with the trends for the published data. We thus conclude that the biomass of phytoplankton in the UMR in general is sensitive to changes in the external supply rate of phosphorus to the water, and we infer that this sensitivity should be true for the majority of MRB sites.

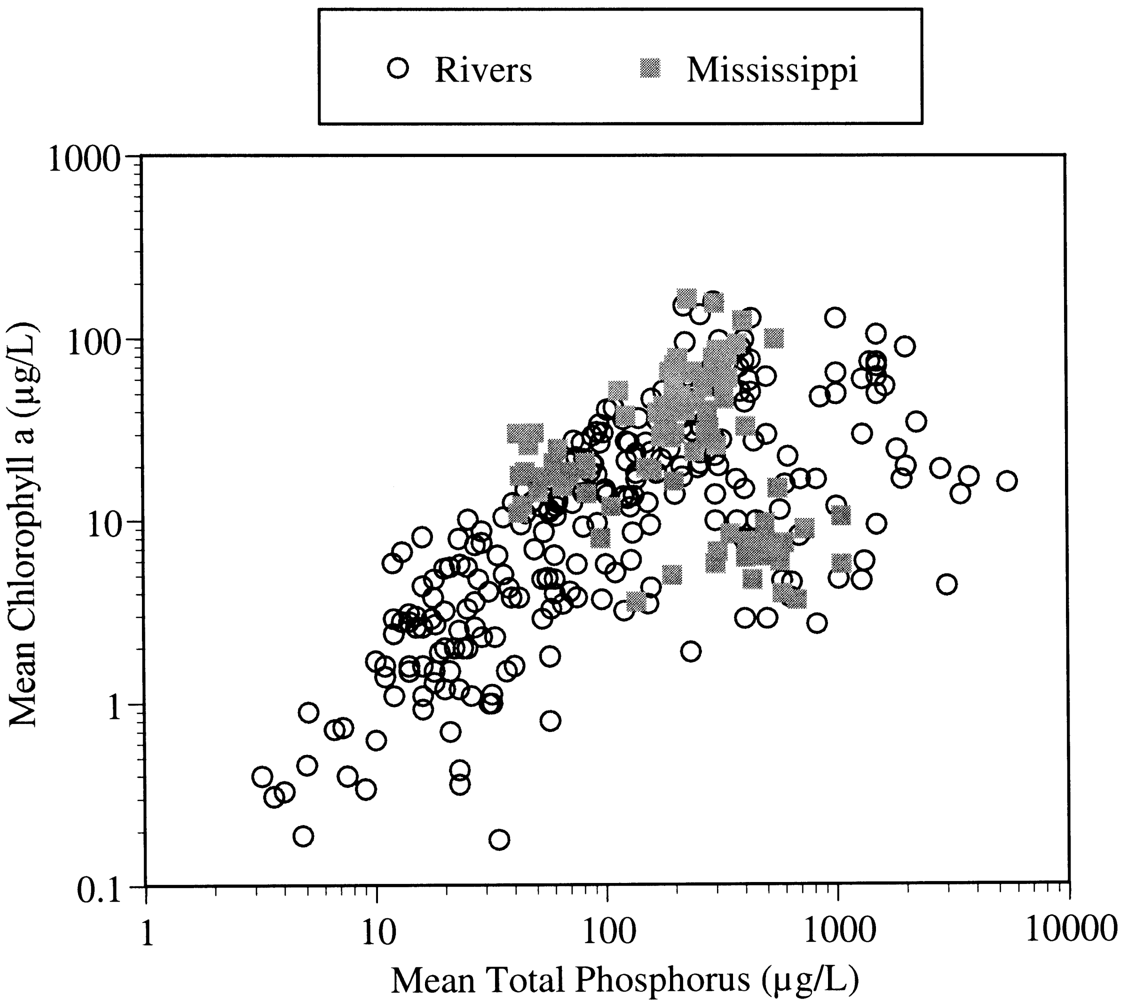


FIGURE 3.29. Growing season mean chlorophyll *a* concentrations versus corresponding TP concentrations for various rivers. *(Data for open circles, from Van Nieuwenhuyse and Jones 1996; data for closed squares, from Metropolitan Council Environmental Services sites in the Twin Cities area.)*

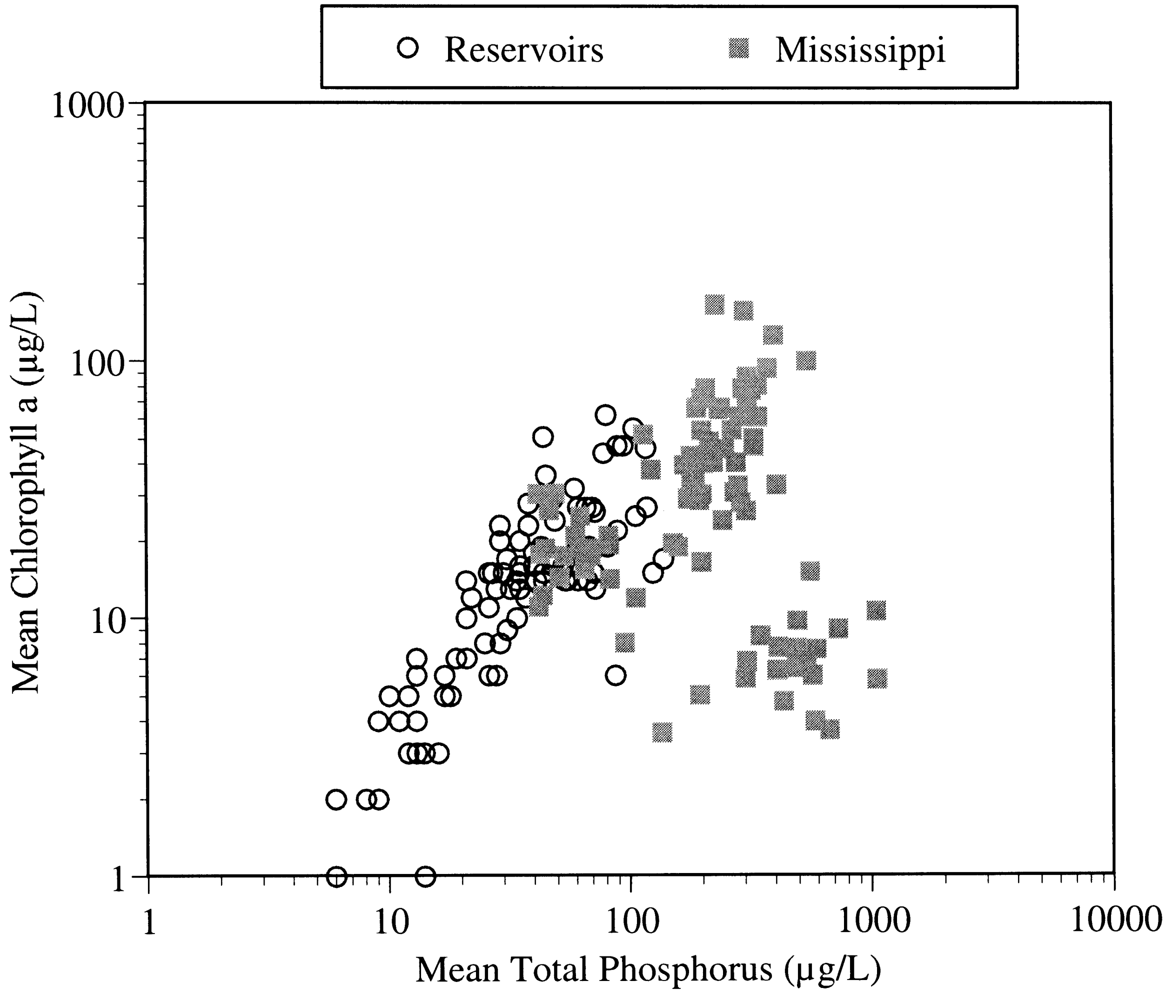


FIGURE 3.30. Growing season mean chlorophyll *a* concentrations versus corresponding TP concentrations for various reservoirs and rivers. *(Reservoir data for open circles, from Jones and Knowlton 1993; data for closed squares, from Metropolitan Council Environmental Services sites in the Twin Cities area.)*

Reduction in the frequency of nuisance algal blooms

Van Nieuwenhuyse and Jones (1996) recently analyzed a large database on chlorophyll and nutrient concentrations from a large number of rivers in the United States and elsewhere. They developed an empirical expression that relates the growing season mean concentrations of suspended chlorophyll *a* (chl *a*, µg/L) in river and streams to their mean concentrations of TP (µg/L) and to the size of the drainage area (catchment size) above the sampling station (A_c, km^2):

$$log\mathrm{Chl} = -1.92 + 1.96 log\mathrm{TP} - 0.30(log\mathrm{TP})^2 + 0.12 log\mathrm{A_c} \quad R^2 = 0.74 \qquad (3\text{-}1)$$

Because a majority of sites in the MRB appear to show potential P limitation based on TN:TP ratios (see earlier discussion), this empirical relationship was used as a preliminary tool to help predict the likely improvements in chl *a* that would occur *on average* following reductions in TP concentrations in river reaches of fixed catchment size. As illustrated in Figures 3.31 and 3.32, the response of chl *a* to reductions in TP is not linear, and a greater improvement is predicted for a given stepwise decrease in TP as a stream becomes less eutrophic. For example, Figure 3.31 shows that an 18% reduction in chl *a* is predicted if stream-water TP is reduced from 125 to 100 µg/L. However, a 52% reduction in chl *a* is predicted for a 25 µg/L decrease in TP if the initial stream-water TP is 50µg/L. A similar trend is shown in Figure 3.32, which illustrates the potential improvements in mean chl *a* that are predicted for 10% reductions in TP.

It is important to recognize that considerable scatter exists in all chl *a*–TP correlations (cf. Figures 3.29 and 3.30; for other aquatic ecosystems, see Reckhow and Chapra 1983 and V.H. Smith 1998). Because many factors other than concentrations of TP can influence the production of algae (and thus the concentration of chl *a* in the water column), one can expect a considerable variation about the actual response of algal biomass to a given change in mean TP concentrations at any given site or water body (for example, see the retrospective study of lake eutrophication by Smith and Shapiro 1981).

In addition, it is important to recognize that responses of chl *a* to reductions in TP concentrations should not be expected to be large when inorganic P values are high and nonlimiting to algal growth. At a site where inorganic P levels are high, very substantial reductions in P loading may be necessary to induce a measurable response in planktonic productivity and biomass.

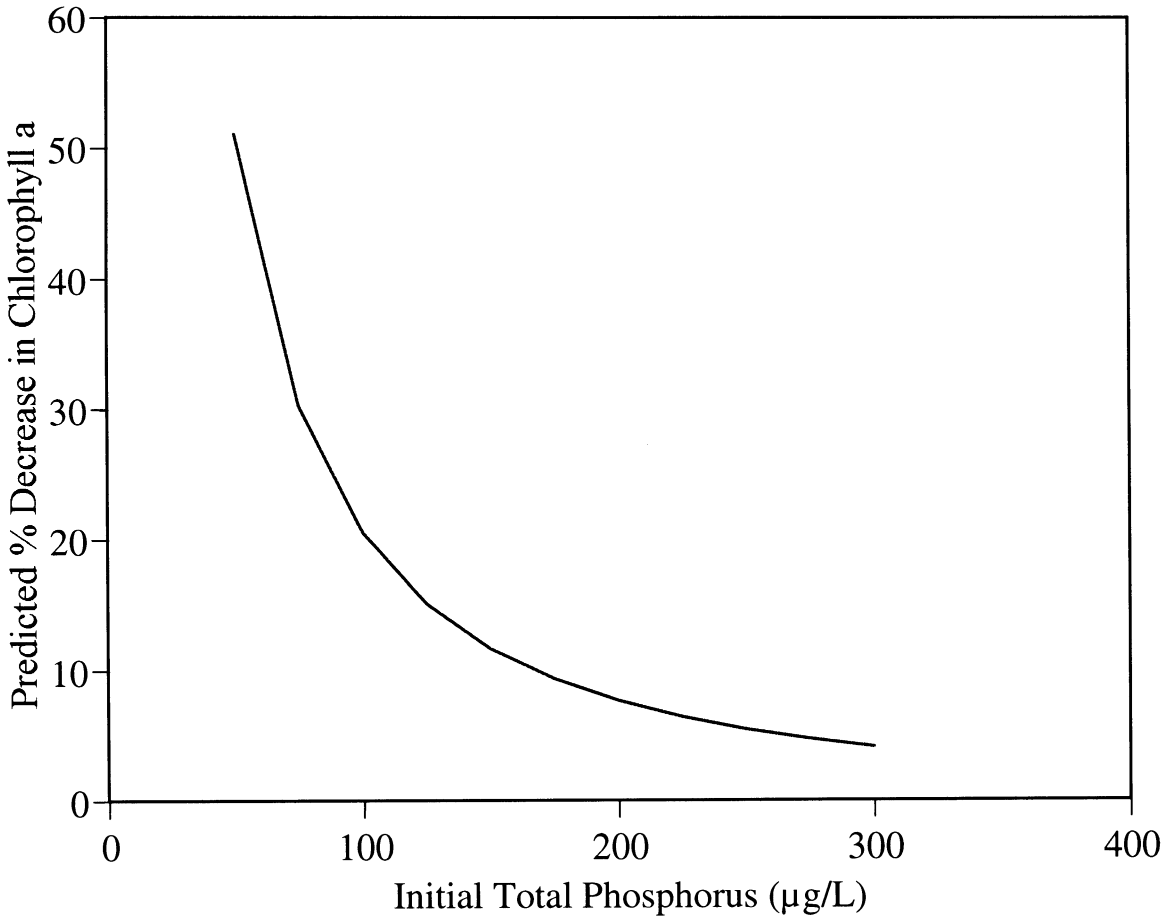


FIGURE 3.31. Predicted percent decrease in growing season chlorophyll *a* concentrations from a 25 µg/L decrease in TP concentrations as a function of initial TP concentrations.

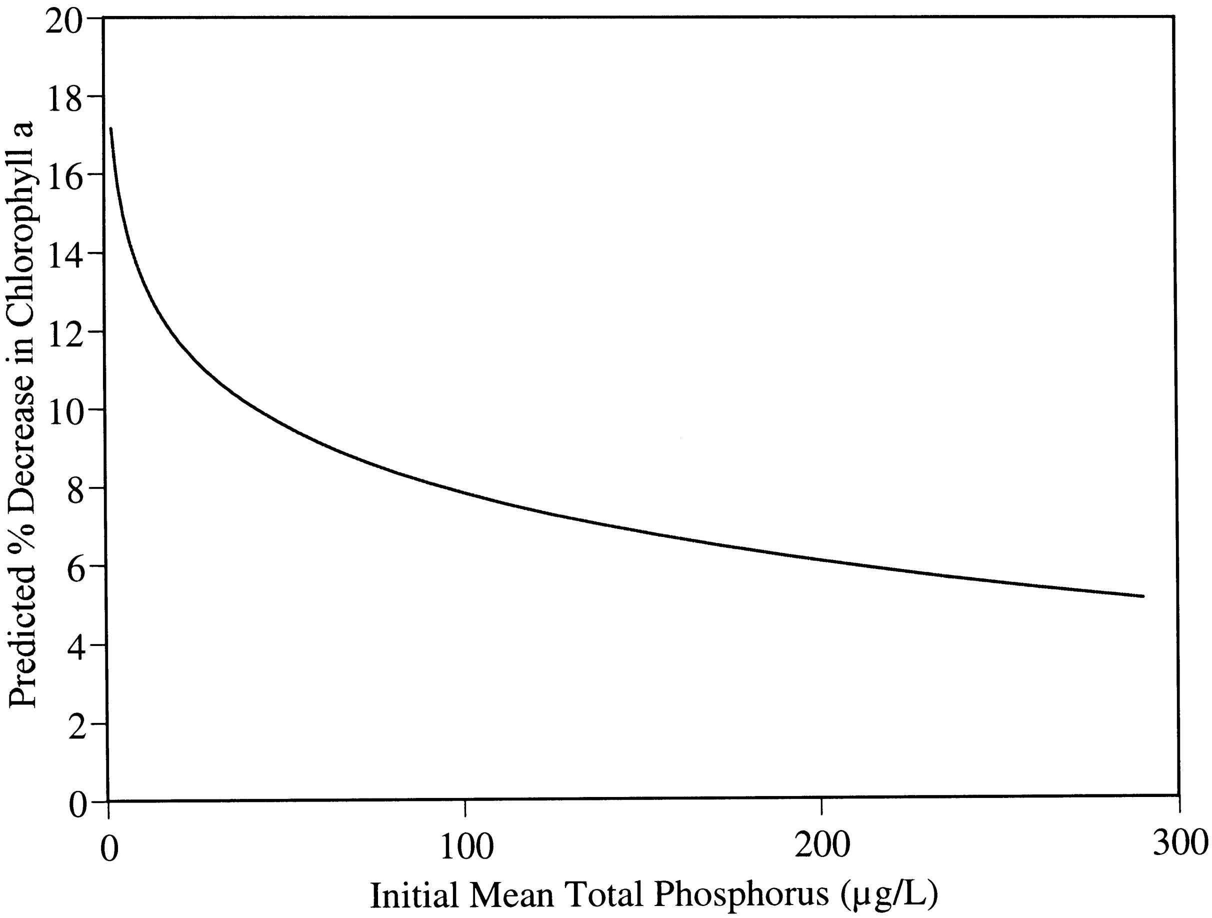


FIGURE 3.32. Predicted percent decrease in growing season chlorophyll *a* concentrations from a 10% decrease in TP concentrations as a function of initial TP concentrations.

Finally, it is useful to note that a strong correlation exists between seasonal peak concentrations of chl *a* and seasonal mean values in the MCES data set (Figure 3.33); peak concentrations are about 2.7 times the mean values in this data set. Consequently, improvements to the mean chl *a* resulting from reductions in mean TP should induce similar responses in peak (nuisance) algal biomass.

Algal species composition

The factors that control the algal composition in flowing waters are not as well understood as those regulating algal biomass. Also, few tools are available to predict the effects of nutrient loading on the community structure of either benthic or suspended algae. For example, efforts have only recently begun to develop algae-based eutrophication indices for rivers in Europe (Kelly 1998). Nonetheless, stream eutrophication does appear to cause shifts in algal species composition (Hynes 1969; Whitton and Rott 1996). In addition, data from a study of the freshwater Potomac (Limno-Tech 1991; V.H. Smith 1998) suggest a possible effect of low TN:TP ratios on cyanobacterial blooms in rivers that parallels the effects observed in freshwater lakes (V.H. Smith 1983). Such N:P ratio effects suggest that it may be important to consider both N and P control measures in the management of eutrophication in flowing waters.

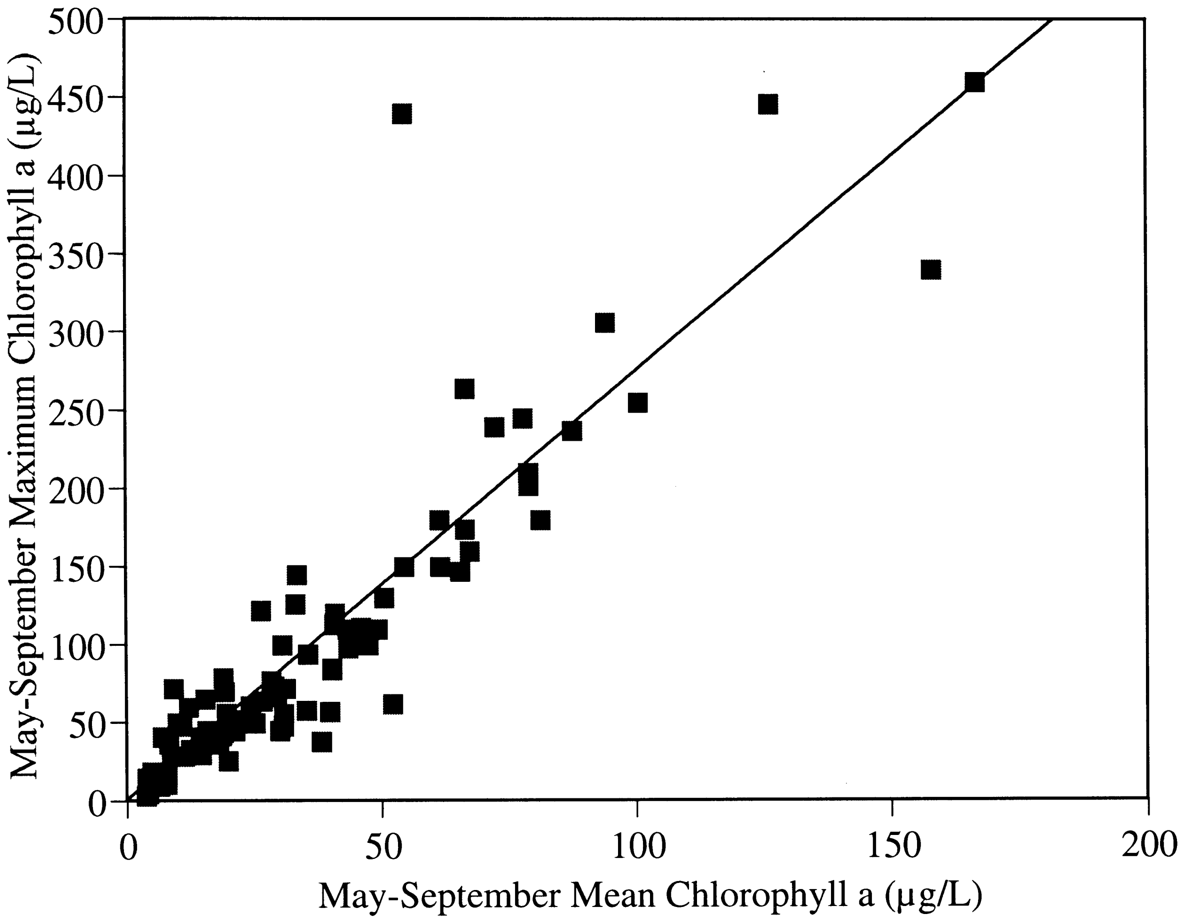


FIGURE 3.33. Maximum chlorophyll *a* concentrations during May–September, versus mean chlorophyll *a* during the same period for seven river sites in the Metropolitan Council Environmental Services data set.

3.2.6.5 MACROPHYTES

The Upper Mississippi River (UMR) supports various forms of aquatic macrophytes, including floating (e.g., duckweed), floating-leaved (e.g., water lilies), emergent (e.g., sedges), and submersed (e.g., wild celery) plants. Among these forms the most likely to be affected by nutrient changes are the floating and submersed macrophytes.

Floating macrophytes derive their nutrients directly and exclusively from the water. They reside only in relatively isolated backwater areas and would be expected to decrease in abundance with decreasing nutrient concentrations. Submersed macrophytes can obtain nutrients from both the water and bottom sediments, but they rely primarily on sediments as their source of nutrients. These macrophytes are most likely to be affected by changes in nonalgal turbidity that may result from land-management changes to decrease nutrient loadings or from decreases in algal turbidity that may result from lower nutrient concentrations in river waters. Emergent and floating-leaved macrophytes are the least likely types to be affected by changes in nutrients in the water because they rely nearly entirely on the bottom sediment for their nutrients. However, they may be affected by reductions in the delivery of sediment nutrients, because of reductions in sediment delivery to the root zone.

Owing primarily to a gradient of increasing turbidity in the UMR, particularly below Pool 13 where tributaries draining predominantly agricultural areas enter the river (Nielsen et al. 1984), submersed macrophytes are most abundant in Pools 1–13 (i.e., from Minneapolis, MN, to just south of Dubuque, IA) (Rogers and Theiling 1999). The depths to which submersed macrophytes occur also indicate the negative relationship between plant abundance and increasing turbidity in a downstream direction within the UMR. Generally, submersed macrophytes are found deeper in the upper pools than in the lower pools (Rogers and Theiling 1999). Below Pool 13, submersed macrophytes are restricted almost entirely to a few hydraulically isolated backwater areas, where turbidity is low. Thus, any changes in nutrients that affect levels of turbidity in the UMR can be expected to affect the abundance and distribution of submersed macrophytes.

Nitrogen and phosphorus are the two nutrients most likely to limit the growth of aquatic macrophytes in locations where demand through growth exceeds supply (Carignan 1985; Barko et al. 1988). Based on results of laboratory and in-situ fertilization studies, N is much more likely than P to limit macrophyte growth under conditions of nutrient deficiency (Anderson and Kalff 1986; Barko et al. 1991; Rogers et al. 1995). These two nutrients are incorporated in plant tissues primarily through root uptake from sediments. Because sediments provide a nutritional subsidy but also decrease underwater light availability, especially when sedimentation is excessive (Barko and James 1998), submersed macrophytes grow best under conditions of intermediate sedimentation.

Nitrogen is a key element in the growth of aquatic macrophytes, and attention to this nutrient for macrophytes needs to be elevated to the same level of importance as P for algae. For the most part, particularly in such nutrient-enriched systems as the UMR, pools of P available for root uptake by macrophytes are substantial. In contrast, pools of sediment N available for macrophyte uptake are much smaller (Carignan 1985; Chen and Barko 1988; Barko et al. 1988). Consequently, N is depleted from sediments much more rapidly than P and is more likely than P to limit macrophyte production. Accordingly, reductions of N in the UMR are likely to have a greater direct effect on macrophyte growth than reductions in P. However, reductions in P that decrease algal growth (both phytoplankton and attached algae on macrophyte leaf surfaces) may improve underwater light availability and consequently increase submersed macrophyte growth.

Aquatic macrophytes have important effects on water-column turbidity, sediment dynamics, and overall water quality in shallow systems. They reduce sediment resuspension and erosion and promote accretion by reducing and/or redirecting turbulent water currents (Fonseca et al. 1982; Gregg and Rose 1982; Madsen and Warncke 1983; Eckman et al. 1989; James and Barko 1994). Aquatic macrophytes also serve as effective sediment traps by intercepting suspended sediment (Patterson and Brown 1979; Wetzel 1979; Carpenter 1981). Macrophytes can inhibit phytoplankton growth directly by releasing allelopathic substances and scavenging nutrients from the water column, and indirectly by providing a refuge for zooplankton grazers (Scheffer et al. 1993). By inhibiting sediment resuspension and reducing algal concentrations, submersed aquatic macrophytes are important regulators of water quality in shallow aquatic systems, including river pools and backwater areas (Carter et al 1988).

In the absence of aquatic macrophytes, shallow-water systems often are dominated by high suspended sediment concentrations induced by wind resuspension and/or benthic fishes. This leads to a variety of sediment-related water quality problems, such as enhanced nutrient cycling, reduced water clarity, and high phytoplankton biomass (Dillon et al. 1990; Maceina and Soballe 1990; Hellström 1991; SØndergaard et al. 1992). The persistence of aquatic macrophytes tends to be associated with clear water and low phytoplankton biomass (Hosper 1989; Dieter 1990; Scheffer 1990). Macrophytes need these conditions to survive, and as described above, their presence in a water body promotes these conditions (an example of positive feedback). Aquatic macrophyte communities thus are critical to water quality in shallow water bodies (Hosper and Jagtman 1990; Hanson and Butler 1994) and are an important factor in river and reservoir restoration (Barko and James 1998).

In summary, if reductions in N and P levels increase underwater light, the distribution of submersed aquatic macrophytes will expand both longitudinally and with depth in the UMR. Effects on emergent and floating-leaved macrophytes are unlikely, except for possible site-specific decreases in productivity if delivery of nutrients to the root zone is reduced. This exception may apply to rooted submersed macrophytes as well. The growth of floating macrophytes may be reduced, but only in the relatively few isolated backwater areas where they reside. Although a large increase in the abundance of aquatic macrophytes has the potential to interfere with recreational uses of the river, an expanded distribution of aquatic macrophytes, particularly submersed forms, generally will be beneficial to UMR water and ecosystem quality, both locally (e.g., in terms of improved habitat for fish) and system-wide (e.g., in terms of increased nutrient retention within the river system). The latter would result from enhanced settling and retention of suspended sediment in macrophyte beds, and this may lead to a significantly lower delivery rate of nutrients to the Gulf of Mexico than that predicted from direct effects of external source reductions alone.

3.2.6.6 Fish Community Composition and Production

Comparative analyses of stream fish communities have revealed patterns in community structure that correlate strongly with trends in stream-water chemistry (e.g., Lyons 1989; Matthews et al. 1992). Because positive correlations between stream-water fertility and fish production in streams have been noted in the literature (e.g., Deegan et al. 1997; Johnston et al. 1990; Hoyer and Canfield 1991), elevated loadings of N and/or P may enhance fish growth at some sites in the MRB. (In this context, nutrient-source reductions could have a negative effect.) However, decreases in water quality related to eutrophication (Table 3.8) may offset these potential increases in fish growth. Moreover, the composition of fish communities changes with increasing degrees of eutrophication toward greater dominance by rough or “trash” fish (shad, bullheads, carp) and lower abundance of game fish (bass, perch, crappies). Sport fishing is more important by far than commercial fishing in the MRB, in terms of both simple economics (monetary value of the activity) and public interest and involvement. Given the fact that rough fish are a common component in fish communities of the MRB, the relative proportions of game and rough fish in MRB fish communities may be a more important issue than total fish production.

Optimal water quality for fisheries thus may occur at low to intermediate levels of stream fertility, rather than in eutrophic or hypereutrophic waters. For example, in a study of Irish rivers dominated by benthic rather than by suspended algae, McGarrigle (1993) concluded that maintaining a mean annual dissolved TP concentration of < 47 µg/L was necessary to prevent the nuisance growth of attached algae and preserve water quality suitable for salmonid fishes. A more recent study by Miltner and Rankin (1998) found negative correlations between nutrients and biotic integrity in Ohio rivers and streams. Deleterious effects of increasing nutrients on fish communities were detectable in low-order streams when total inorganic N and TP concentrations exceeded 0.61 mg/L and 0.06 mg/L, respectively. These concentrations are far lower than those found in a majority of cataloging units sampled in the MRB (see section 3.2.6.3). Therefore, if reductions in nutrient loading are sufficient to produce general improvements in trophic-state conditions in rivers of the MRB, one also might expect improvements in the river system’s biotic integrity and fish-related water quality.

The task group draws this conclusion with caution, however. Our knowledge of the links between nutrients and fisheries in streams is still limited, and there may be significant site-to-site or regional variability in this relationship. For example, in temperate warm-water streams where the food web is typically more complex than in cold-water streams, the trophic responses to nutrient enrichment may be indirect and difficult to predict (Johnston et al. 1990). Because no satisfactory framework exists for predicting the effects of eutrophication on fish species composition or on game fish production in streams and rivers, we can make no quantitative statements concerning the potential beneficial effects of decreases in nutrient loading on fisheries in the MRB. Nonetheless, it is important to note that a recent study by Eklöv et al. (1998) documented significant responses of stream fish assemblages to improved water quality in southern Sweden.

3.2.6.7 CONCLUSIONS: ECOLOGICAL EFFECTS OF NUTRIENT-SOURCE REDUCTIONS

Overall, the ecological and water quality effects of lower nutrient loadings to an aquatic system that generally can be regarded as "overenriched" can be expected to be positive. This statement certainly applies to flowing waters in the MRB. As described in the previous section, the benefits of nutrient-source reduction in the MRB include: lower concentrations of the nutrients, potentially limiting algal growth in the waters; decreased frequency of violations of nutrient-related water quality standards; decreased frequency of nuisance algal blooms; higher water clarity; increased abundance of submerged aquatic macrophyte communities; and possible improvements in the structure of fish communities. The exact magnitude and geographic distribution of such improvements are impossible to predict without a much more detailed analysis of the nature, magnitude, and geographic distribution of the changes in land-management practices and other nutrient-source reductions.

3.2.7 Potential Negative Effects of Nutrient-Source Reductions

Although the beneficial effects of nutrient reduction on waters of the MRB are numerous (as described above), the task group also considered the potential for negative ecological impacts of nutrient-source reductions on MRB aquatic ecosystems. The panel identified three possible negative effects: (1) decreased losses of nitrogen in the system by denitrification, as a result of lower nitrate concentrations; (2) decreased rates of photochemical degradation of organic contaminants by hydroxyl radicals (•OH), also because of lower nitrate concentrations (nitrate is the primary source of photo-generated •OH in most aquatic systems); and (3) decreased fish production. The first two are minor issues; the third is a potentially important issue, but as described in section 3.2.6.6 and further in this section, it is unlikely to be a significant problem in the MRB under any reasonable scenario for nutrient reduction.

If water-column nitrate is the only source of nitrate lost by denitrification in MRB river sediments (i.e., if mass transport of nitrate across the water-sediment interface is the only mechanism for denitrification in the river), then any decreases in nitrate concentrations in the water column should cause a proportionate decrease in denitrification. This is because the mass transport of nitrate across the water-sediment interface is a first-order process (i.e., directly proportional to nitrate concentration). Although the net effect would be to decrease the total mass of nitrate lost within the river system, it would not change the fraction of nitrate in the system that would be lost. Assuming that other factors remain constant, if 20% of the nitrate in the river is lost by denitrification under present conditions, 20% of the nitrate in the river should also be lost even if nitrate levels decline because of nutrient-source controls. (This assumes that the longitudinal distribution of nitrate concentrations in the rivers would not be affected by nutrient-source controls.)

Mineralization and nitrification of organic N in the aerobic surficial sediment layer represent unknown but probably significant sources of the nitrate lost by denitrification in the anoxic sediment layer (e.g., Seitzinger 1988; see section 3.2.5.5). Because concentrations of organic N in river sediments probably are related only loosely to N concentrations in the water column and to N inputs to the river, it is likely that the amount of N lost by the coupled nitrification–denitrification process would not change much under moderate shifts in nutrient inputs. Consequently, it is likely that decreases in N inputs to the MRB system as a result of changes in land-management practices would result in a less than proportional decrease in denitrification losses within the river system.

For many decades, nitrate was considered to be chemically unreactive in aquatic systems, and essentially all its transformations were thought to be biologically mediated. Over two decades ago, aquatic chemists discovered that nitrate is photochemically reactive and that nitrate photolysis results in the formation of highly reactive hydroxyl radicals (•OH). More recent evidence (Zepp et al. 1987) indicates that nitrate is the primary source of •OH radicals in surface waters. These radicals react rapidly with both anthropogenic organic contaminants (e.g., pesticides, polychlorinated biphenyls (PCBs), polynuclear aromatic hydrocarbons (PAHs), and natural organic matter in water, causing their (indirect) photochemical degradation. Rate constants for reaction of •OH with a wide variety of organic compounds are near the limits of diffusion control (~10^9-10^{10} M^{-1} s^{-1}); rate constants for reaction of •OH with natural organic matter typically are about 2-3 x 10^4 (mg C/L)$^{-1}$ s^{-1} (Brezonik and Fulkerson–Brekken 1998). To the extent that nitrate concentrations decrease in the rivers as a consequence of improved nutrient-source controls, it is reasonable to predict that photochemical production of •OH also will decline. Countering this prediction, however, is the fact that nitrate photolysis depends on light intensity, and reductions in nutrient concentrations may improve water clarity and the depth of the water column accessible to sunlight. Consequently, the decrease in photolysis resulting from lower nitrate levels and the increase in rates because of higher light availability may balance out. Moreover, indirect photolysis is only one of several loss mechanisms for organic contaminants, and for many contaminants it is less important than hydrolysis, biodegradation, volatilization, or loss by sediment deposition. Finally, it can be argued that the organic contaminants should not be in the water anyway, and that it is unreasonable to expect that humans can rely on the presence of another contaminant (nitrate) to remove them from aquatic systems.

Regarding the relationship between nutrient levels and fish production, it first should be noted that only the crudest correlations exist; a wide range of fish-management practices also influences the total fish production of a given aquatic system. Furthermore, because the rivers (and reservoirs) of the MRB are managed as sport fisheries, the total production of fish biomass is not nearly as important as the production of game fish in numbers and sizes desired by anglers. Many factors besides (and in many case more important than) N and P loading rates influence these variables. Moreover, projected changes in nutrient inputs to MRB rivers from improved land-management practices and additional controls on point sources are not likely to have dramatic effects on trophic-state conditions in the rivers; i.e., they most likely will remain moderately productive systems. On this basis, it seems unlikely that sport fisheries would be affected negatively. Any declines that may occur in total biomass production most likely would be compensated for by improvements in habitat and other changes that would promote the development of game fish populations over rough fish populations.

3.3 GULF OF MEXICO

This section contains results for the Gulf of Mexico portion of this study. Section 3.3.1 describes the approach used to generate forecast simulations with the Nutrient Enhanced Coastal Ocean Productivity (NECOP) water quality model. Section 3.3.2 describes the model's principal assumptions and additional assumptions underlying the forecast simulations. Section 3.3.3 presents results of the forecast simulations in the form of comparisons among different years and boundary conditions for both N and P reductions. Section 3.3.4 presents results of sensitivity analyses for different chemical–biological process mechanisms in the model. Finally, Section 3.3.5 discusses results from the forecast simulations and the sensitivity analyses.

3.3.1 Approach to Forecasting Simulations

The calibrated NECOP water quality model was run for a series of forecast simulations. These simulations involved a range of reductions from 10% to 70% in N and P loadings from the Mississippi–Atchafalaya River (MAR). This range was not necessarily intended to represent the range of loading reductions that may be feasible in terms of technology, economics, or social acceptability. Rather, its purpose was to investigate whether loading reductions of 20–30% were sufficient to produce a water quality response, or whether reductions of up to 70% may be required to produce a response. With respect to technical achievability, Task Group 5 (Mitsch et al. 1999) concluded that the N loading to the Gulf of Mexico could be reduced by more than 50% by implementing a number of proven techniques working in concert.

Before any mass-balance model can conduct forecast simulations, the model inputs must be determined and predicted themselves. Because making absolute predictions of MAR inflows, meteorological conditions, or nutrient loadings is impossible, using a mass-balance model to make absolute predictions of the future is also impossible. Results from forecast simulations can provide useful information on trends and approximate magnitudes of system responses under a specified set of assumptions. Mass-balance models are most useful for comparing responses among different possible future scenarios.

To address uncertainties due to potential differences in environmental conditions, separate forecast simulations were conducted for July 1985, August 1988, and July 1990 for each load reduction. To address uncertainties in specification of external boundary conditions, each load-reduction simulation was conducted under two assumptions: (1) all seaward and sediment boundary conditions held constant at base-calibration values, and (2) all seaward and sediment boundary conditions reduced by the same percentage as the nutrient loading in each simulation.

Seaward boundary conditions included nutrient, chlorophyll, dissolved oxygen, and carbonaceous biological oxygen demand (CBOD) concentrations. Sediment boundary conditions included sediment oxygen demand (SOD) and sediment–water nutrient fluxes. In the case of seaward boundary conditions for dissolved oxygen, the oxygen deficit between the base-calibration value and the dissolved oxygen saturation value was reduced, not the dissolved oxygen boundary concentration itself.

The rationale for two different assumptions on boundary conditions was twofold: (1) these forcing functions are not computed by the model, but must be externally specified using available field data; and (2) values for these functions are not independent of MAR nutrient loadings, but can be expected to decrease as MAR loadings decrease. This approach was intended to bracket results of the forecast simulations between present conditions and estimates of future conditions for these forcing functions. The necessity to bracket results between two limiting assumptions on seaward boundary conditions is a consequence of the model's limited spatial domain. The seaward boundaries are not far enough removed to be independent of sources of MAR nutrient loadings. The necessity to bracket results between different sediment conditions arises because the model does not explicitly represent dissolved oxygen or nutrient processes in the sediments.

3.3.2 Assumptions

Results of the forecast simulations in this report are premised on all of the assumptions inherent in the calibrated NECOP model, as well as on additional assumptions contained in the forecast simulations themselves. The calibrated model has the following principal assumptions: (1) the actual environmental system is fully represented by the model's conceptual framework, including model state variables, governing equations, process mechanisms, and external forcing functions; (2) N and P are the only nutrients that potentially limit primary productivity; (3) the actual environmental system is represented only at the coarse spatial scale of the model's segmentation grid (i.e., near-field gradients in the vicinity of the Mississippi and Atchafalaya River plumes, and hypoxia in near-bottom waters, are not explicitly represented); and (4) the actual environmental system is represented in terms of a single "snapshot" in time, corresponding to an assumed summer average, steady-state period (i.e., the potential influences of meteorological events, shelf-edge upwellings, and mesoscale shelf circulation are not explicitly represented).

The forecast simulations have two additional assumptions: (1) all forecast results are estimates of future states of the system, and do not contain any information on the time frame required for the system to fully respond to imposed changes in nutrient loadings; and (2) all forecast results for reduced boundary conditions assume that seaward and sediment boundary conditions will eventually change by the same percentage as the corresponding imposed changes in nutrient loadings. The limitations imposed by the first assumption stem from the fact that the NECOP model was applied in a steady-state mode and, in its present form, cannot be used to forecast response-time trajectories for water quality parameters on the Louisiana Inner Shelf. The limitations imposed by the second assumption are due to the model's restricted spatial domain and the lack of a sediment submodel to represent SOD and sediment–water nutrient fluxes.

3.3.3 Results of Forecasting Simulations

The principal water quality response parameters were bottom-water dissolved oxygen concentrations and surface-water chlorophyll concentrations. Results are presented for both constant and reduced boundary conditions. Forecast results are organized in terms of comparisons for different years, different response parameters, and loading reductions for different nutrients. All comparisons are made using the average of dissolved oxygen concentration responses for individual bottom offshore model segments (Segments 15–21) and the average of chlorophyll concentration responses for individual surface offshore segments (Segments 8–14). Forecast results also are presented that compare responses among different individual spatial segments along the shelf bathymetry. In all cases, results are expressed in terms of changes relative to base-calibration results, not on the absolute values of the forecasts.

All forecast results for dissolved oxygen are based on volumetric concentrations in the bottom offshore segments of the model's spatial segmentation grid. Neither near-bottom hypoxia nor the areal extent of hypoxic waters is explicitly represented. An attempt was made to use the available shelfwide data to develop relationships between volumetric concentrations at the scale of the model's segmentation grid and the areal extent of hypoxia. Because of the large degree of spatial heterogeneity in hypoxic water masses, it was impossible to develop significant quantitative relationships. Consequently, it is impossible to directly translate volumetric dissolved oxygen concentrations computed by the model to areal extent of hypoxia.

3.3.3.1 INFLUENCE OF SEAWARD BOUNDARY AND HYDROLOGICAL CONDITIONS

Forecast responses of average dissolved oxygen and chlorophyll concentrations to nutrient-loading reductions depend heavily on assumptions for boundary conditions. For example, in response to 50% N-loading reductions for 1985 conditions, average dissolved oxygen concentrations increase by less than 5% for constant boundary conditions and by 45% for reduced boundary conditions (Figure 3.34, top panel). For the same forecast simulations, average chlorophyll concentrations decrease by approximately 5% for constant boundary conditions and by 35% for reduced boundary conditions (Figure 3.34, bottom panel).

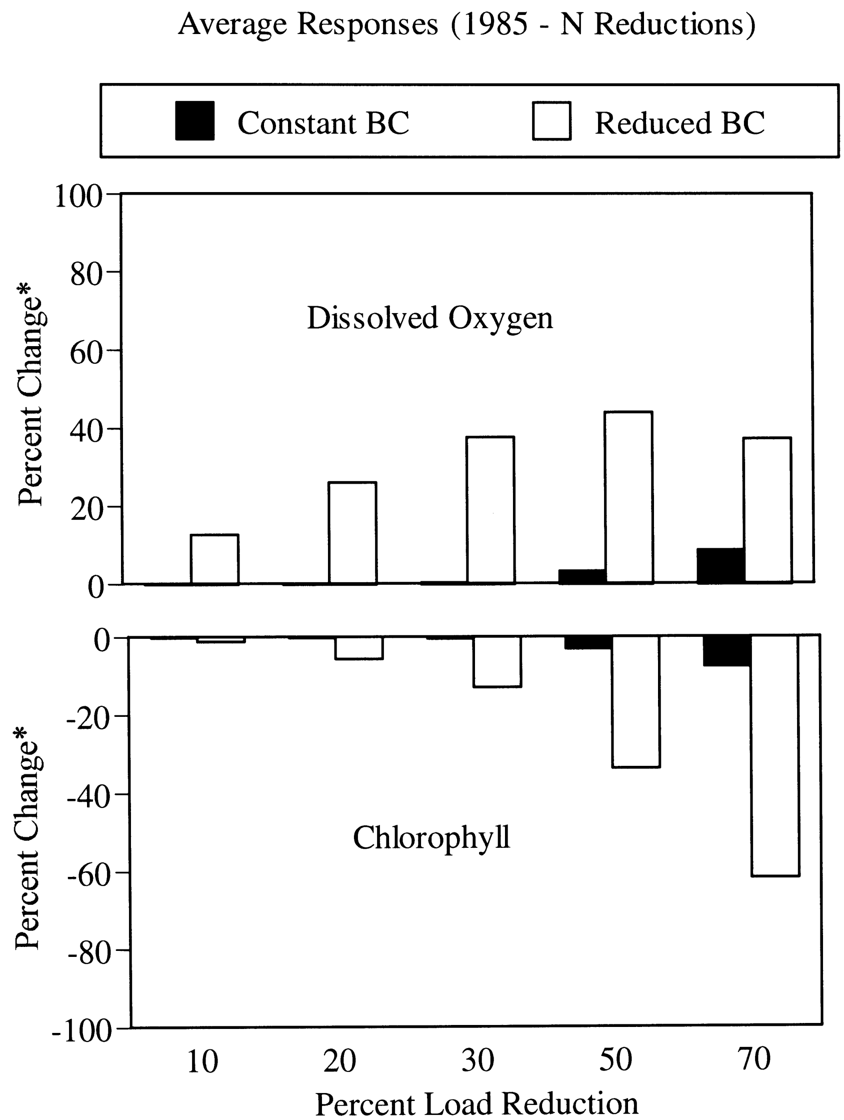


FIGURE 3.34. Predicted responses of average dissolved oxygen and chlorophyll concentrations to nitrogen-loading reductions for 1985 conditions under constant and reduced boundary conditions (BCs).

Differences in responses of average dissolved oxygen concentrations between constant and reduced boundary conditions are not constant among different years. For example, in response to 50% N-loading reductions for 1990 conditions, average dissolved oxygen concentrations increase by 10% for constant boundary conditions and by 140% for reduced boundary conditions (Figure 3.35, top). For the same forecast simulations, differences in responses of average chlorophyll concentrations due to differences in boundary conditions (Figure 3.35, bottom panel) are approximately the same as the corresponding differences for 1985 conditions (Figure 3.34, bottom).

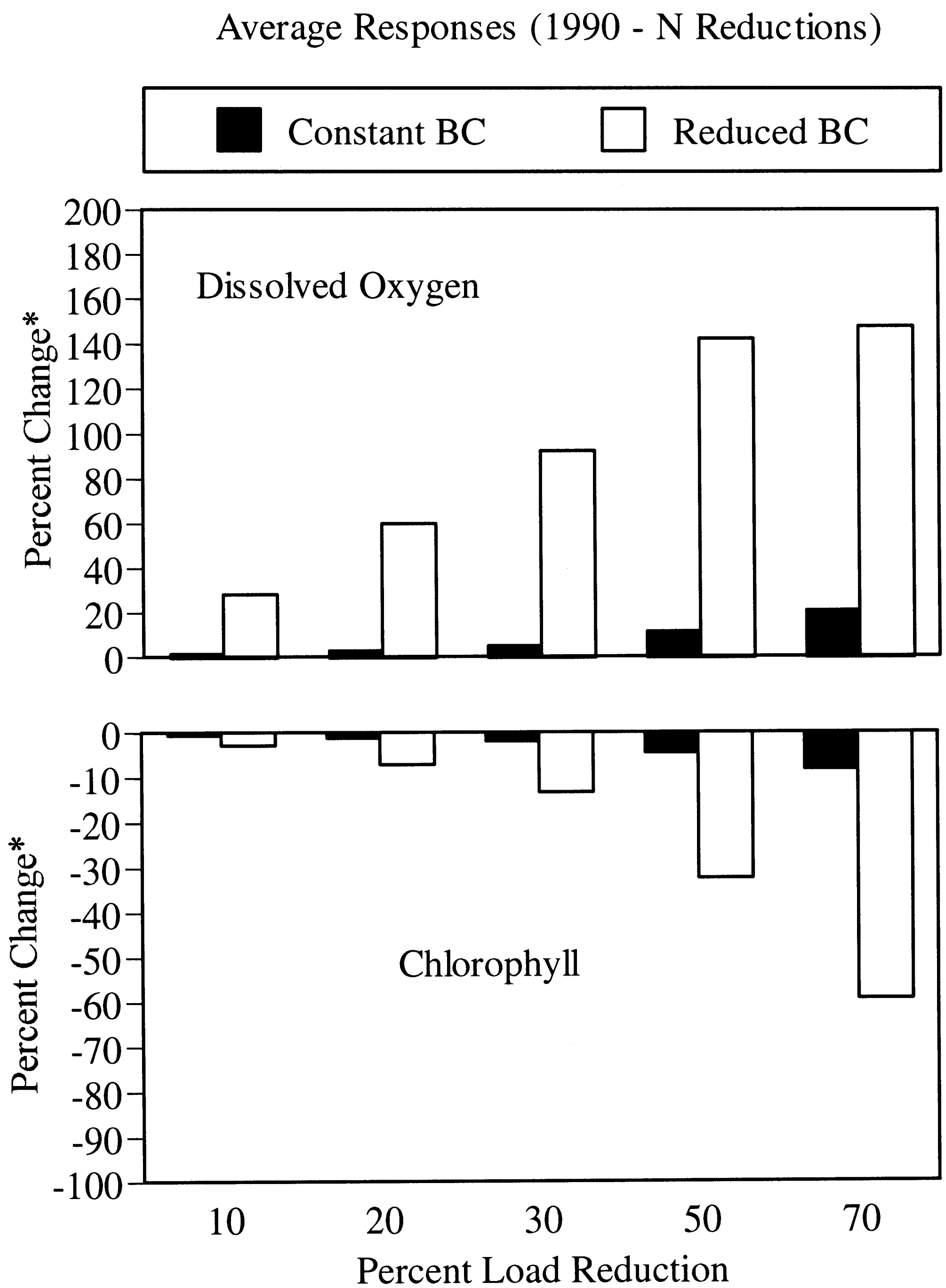


FIGURE 3.35. Predicted responses of average dissolved oxygen and chlorophyll concentrations to nitrogen-loading reductions for 1990 conditions under constant and reduced boundary conditions (BCs).

3.3.3.2 INFLUENCE OF SEDIMENT BOUNDARY CONDITIONS

Responses of average dissolved oxygen concentrations are more sensitive to differences in SOD than to differences in any other boundary conditions. Comparison of the results in the top panel of Figure 3.36 with those in Figure 3.37 indicates that differences in responses due to reductions in SOD are much larger than differences due to reductions in dissolved oxygen deficit values at

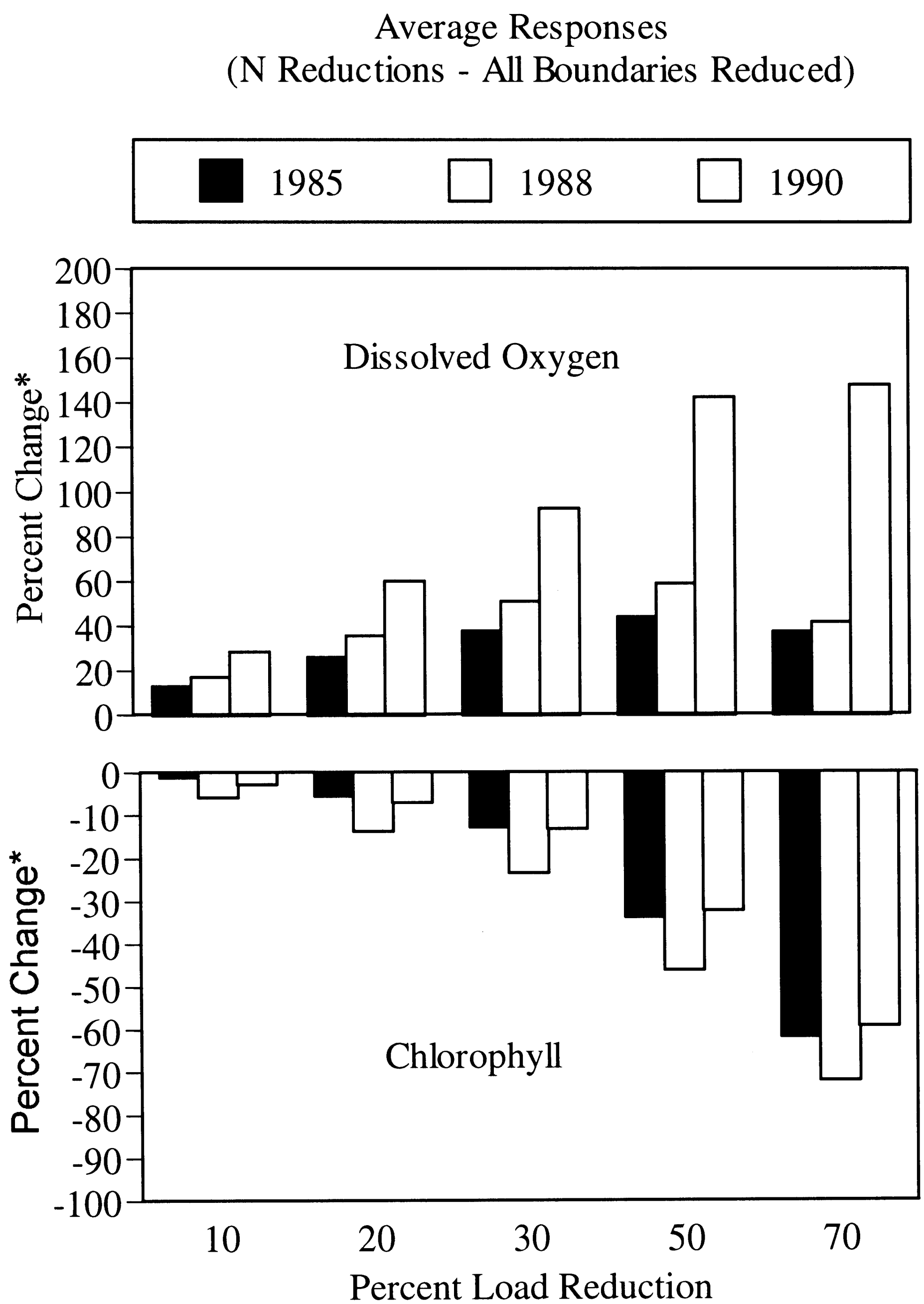


FIGURE 3.36. Predicted responses of average dissolved oxygen and chlorophyll concentrations to nitrogen-loading reductions for 1985, 1988, and 1990 conditions under reduced boundary conditions.

seaward boundaries. In the top panel of Figure 3.37, all boundary conditions were reduced, except for dissolved oxygen deficit values at the seaward boundaries. In the bottom panel, all boundary conditions were reduced, except for SOD. Responses of average dissolved oxygen concentrations for fully reduced boundaries (Figure 3.36, top panel) are due mostly to reductions in SOD (Figure 3.37, top panel), as opposed to reductions in dissolved oxygen deficits at seaward boundaries (Figure 3.37, bottom panel).

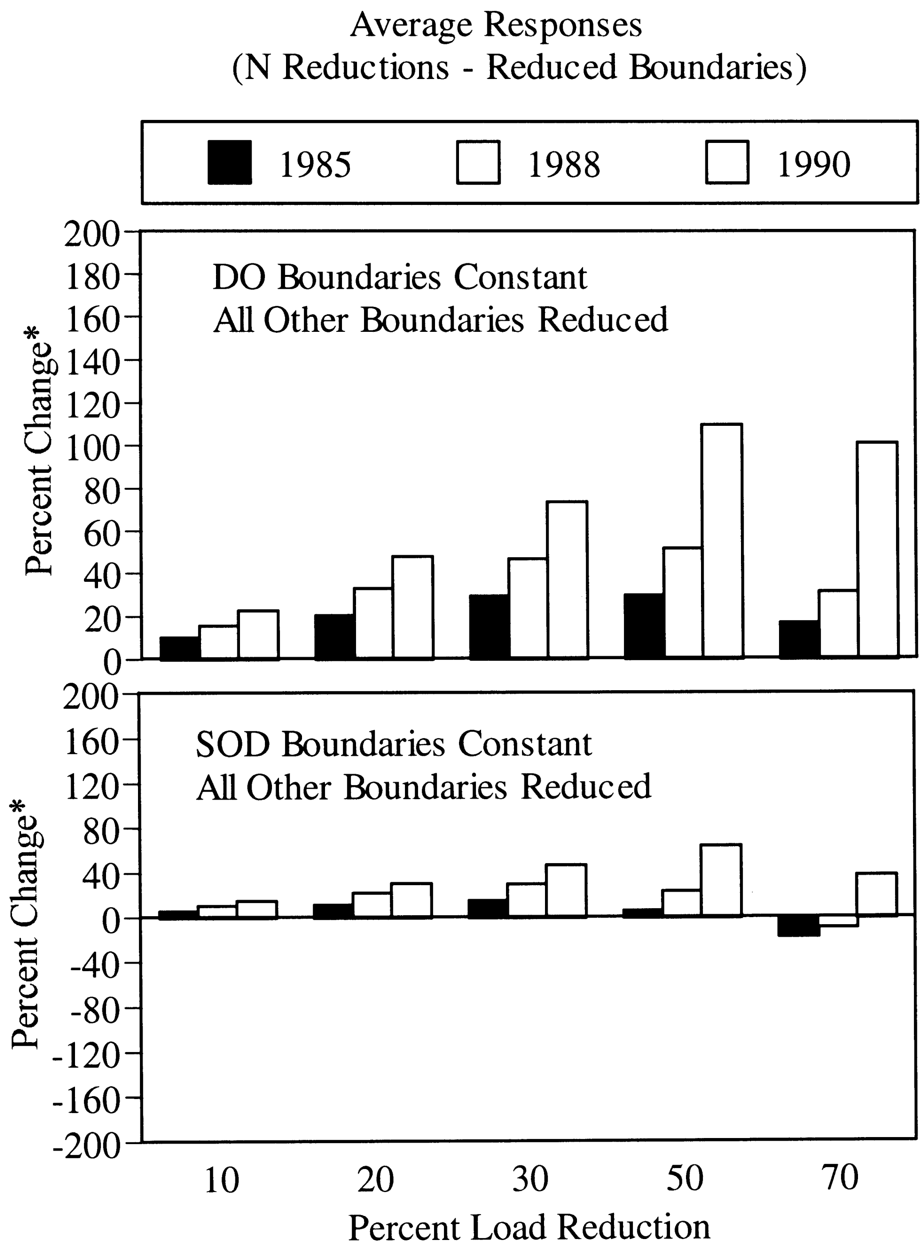


FIGURE 3.37. Predicted responses of average dissolved oxygen concentrations to nitrogen-loading reductions for different assumptions on seaward and bottom boundaries for 1985, 1988, and 1990 conditions.

3.3.3.3 DIFFERENCES BETWEEN NITROGEN AND PHOSPHORUS REDUCTIONS

Differences in results between reductions in N and P loadings were generally not significant, although there was a tendency for responses to be somewhat greater for reductions in N loadings than reductions in P loadings. The largest differences occur for responses of average dissolved oxygen concentrations under reduced boundary conditions. These differences can be illustrated using results for 1985 conditions (Figure 3.38). For this case, maximum increases in average dissolved oxygen concentrations are 45% and 30%, respectively, for N and P reductions.

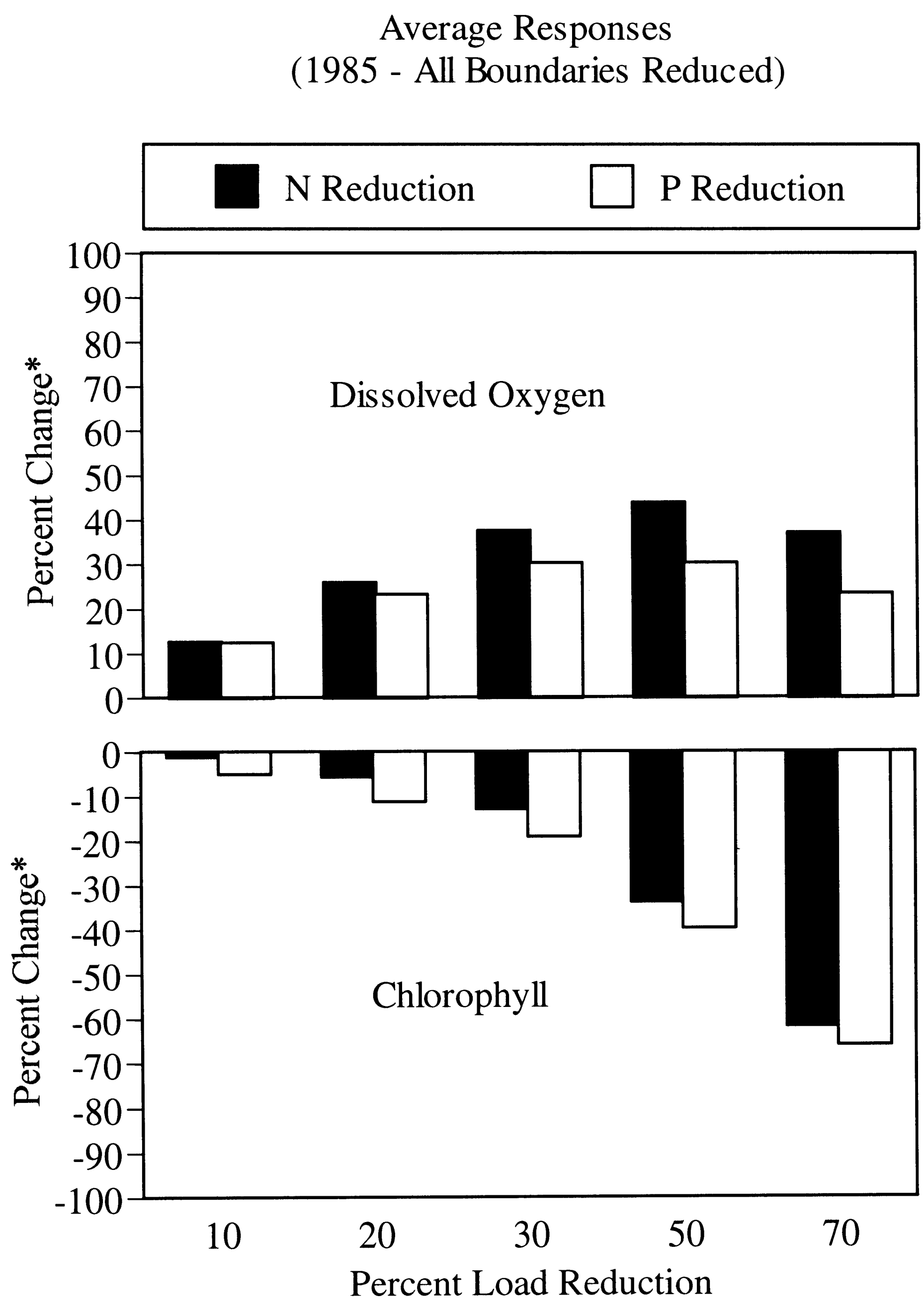


FIGURE 3.38. Predicted responses of average dissolved oxygen and chlorophyll concentrations to nitrogen- and phosphorus-loading reductions for 1985 conditions under reduced boundary conditions.

There was no evidence of significant interactions between reductions in N and P loadings in the forecast simulations. Results of simulations in which N and P loadings were reduced simultaneously were generally consistent with results of simulations in which the more limiting of the two nutrients was reduced by itself. That is, if N was more limiting than P for a particular load reduction and set of boundary conditions, then results for this simulation were not significantly different when N and P loadings were reduced simultaneously by the same percentage.

3.3.3.4 RELATIVE MAGNITUDES OF OXYGEN AND CHLOROPHYLL RESPONSES

For 1985 conditions and reduced boundary conditions, average chlorophyll concentrations are less responsive than average dissolved oxygen concentrations at intermediate (10–30%) and more responsive at higher (50–70%) N-loading reductions (Figure 3.39, top). Differences in responses for 1988 conditions (not shown) follow patterns very similar to those for 1985 conditions.

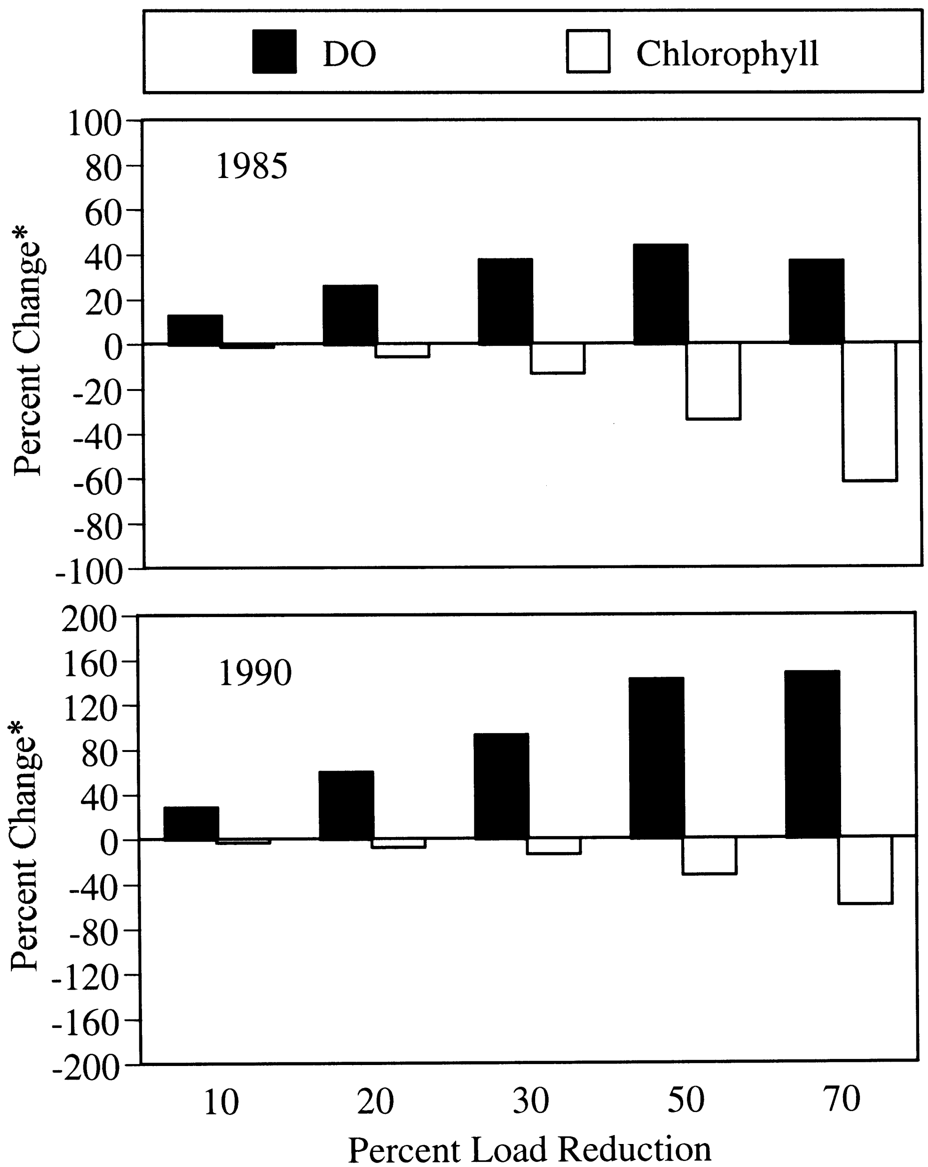


FIGURE 3.39. Predicted responses of average dissolved oxygen and chlorophyll concentrations to N-loading reductions for 1985 and 1990 under reduced boundary conditions.

In contrast to results for 1985 and 1988 conditions, responses of average dissolved oxygen for 1990 are much greater than average chlorophyll responses for a given N-loading reduction under reduced boundary conditions (Figure 3.39, bottom panel). These differences occur across the entire range of N-loading reductions from 10% to 70%. Differences in maximum responses between these two cases are + 150% (dissolved oxygen) and –70% (chlorophyll).

3.3.3.5 DIFFERENCES AMONG SPATIAL REGIONS

For 1985 conditions and reduced boundary conditions, average dissolved oxygen concentrations show much larger responses at greater distances from the Mississippi Delta for intermediate (10–30%) N-loading reductions (Figure 3.40, top panel). However, results for different spatial segments

tend to converge at higher (50–70%) N-loading reductions. For the same simulations, average chlorophyll concentrations show a tendency to be larger at greater distances from the delta (Figure 3.40, bottom panel). Differences in responses for 1988 conditions (not shown) follow patterns very similar to those for 1985 conditions.

In sharp contrast to results for 1985 and 1988 conditions, responses of average dissolved oxygen concentrations under reduced boundary conditions (Figure 3.41, top panel) are much greater near the delta at higher N-loading reductions for 1990. For the same simulations, average chlorophyll concentrations show much larger responses at greater distances from the delta (Figure 3.41, bottom panel).

3.3.4 Sensitivity Analysis

To address uncertainties due to representation of chemical–biological processes, the task force conducted a series of sensitivity analyses with the calibrated model. All internal model process rates, stoichiometric coefficients, and SOD were varied by plus and minus 30%, and results were compared to base calibration results. These sensitivity analyses were not designed to represent actual uncertainties in each parameter, but rather to determine the relative sensitivity of model results to systematic variations across different parameters.

Results are presented for only the five processes to which model responses were most sensitive. These include variations in the underwater light-extinction coefficient, saturation light intensity, the carbon:chlorophyll ratio, water-column oxygen demand (CBOD decay rate), and SOD. Model responses were also sensitive to variations in nitrogen:carbon (N:C) and phosphorus: carbon (P:C) stoichiometric ratios for the phytoplankton, and nutrient mineralization rates. Results are presented for 1985 (Figure 3.42) and 1990 (Figure 3.43) conditions. Results for 1988 conditions were not substantially different from those for 1985.

Dissolved oxygen concentrations were more sensitive to variations in light-extinction coefficients and saturation light intensities than to variations in any other process parameters. Chlorophyll concentrations were sensitive to variations in light-extinction coefficients, saturation light intensities, and carbon:chlorophyll ratios, but were not sensitive to variations in the CBOD decay rate or SOD. Dissolved oxygen concentrations were more responsive under 1990 conditions than under 1985 conditions; however, the differences in chlorophyll concentration responses between these two years were insignificant. In general, model responses were not symmetric about equal plus-and-minus variations in model parameters, especially for variations in light-related parameters.

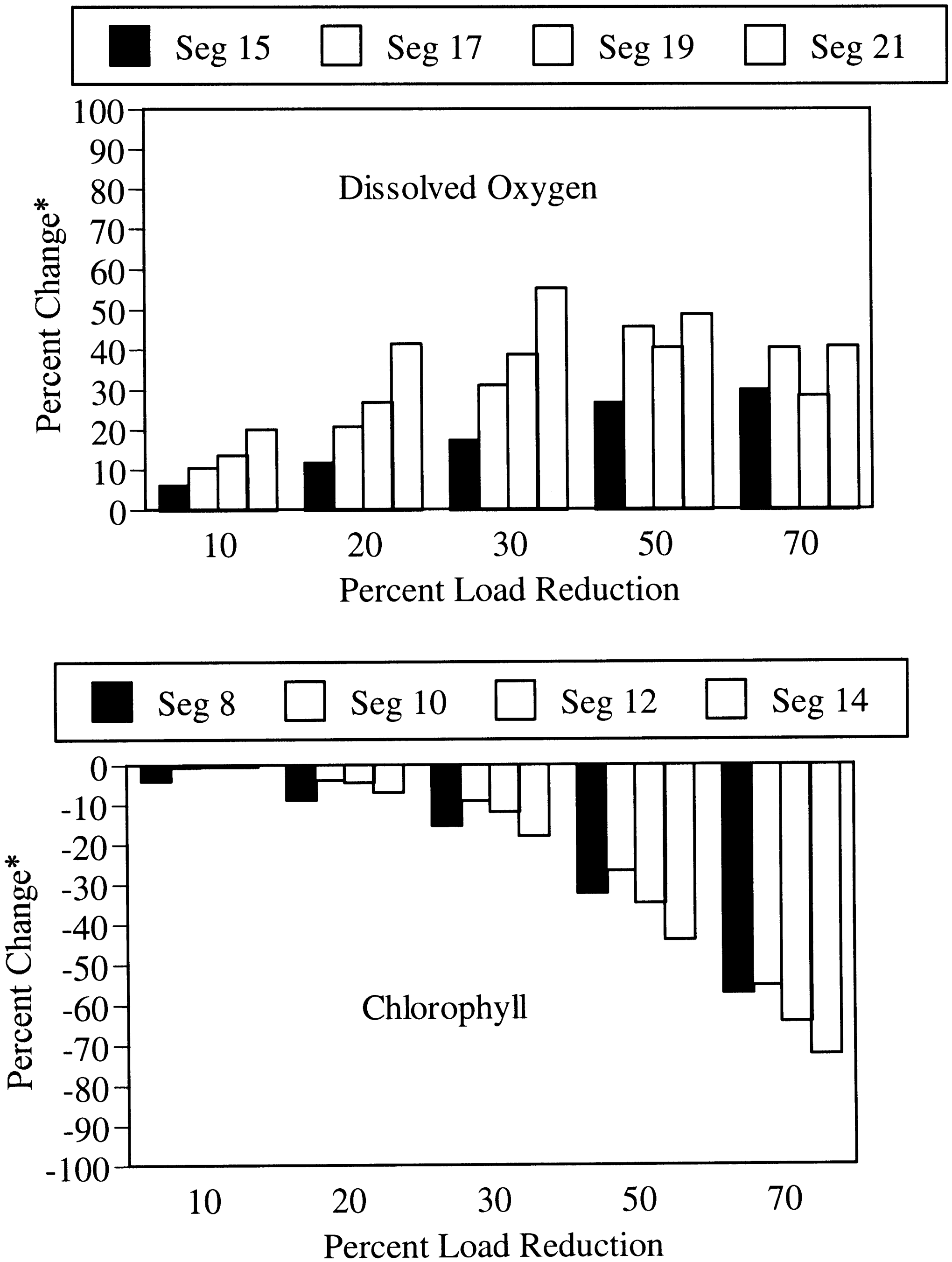


FIGURE 3.40. Predicted responses of average dissolved oxygen and chlorophyll concentrations for different spatial segments of nitrogen-loading reductions for 1985 conditions under reduced boundary conditions.

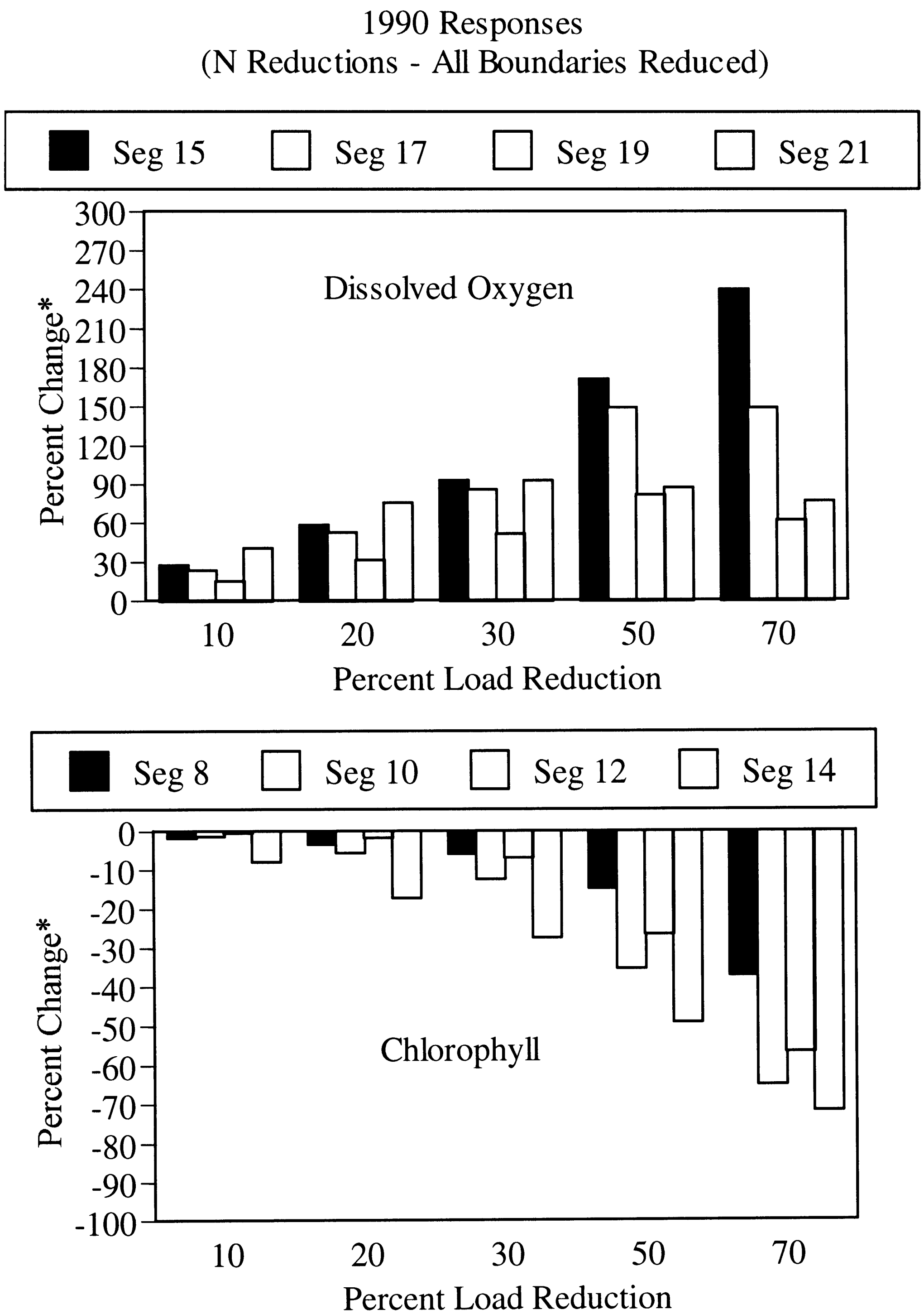


FIGURE 3.41. Predicted responses of average dissolved oxygen and chlorophyll concentrations for different model spatial segments to nitrogen-loading reductions for 1990 conditions under reduced boundary conditions.

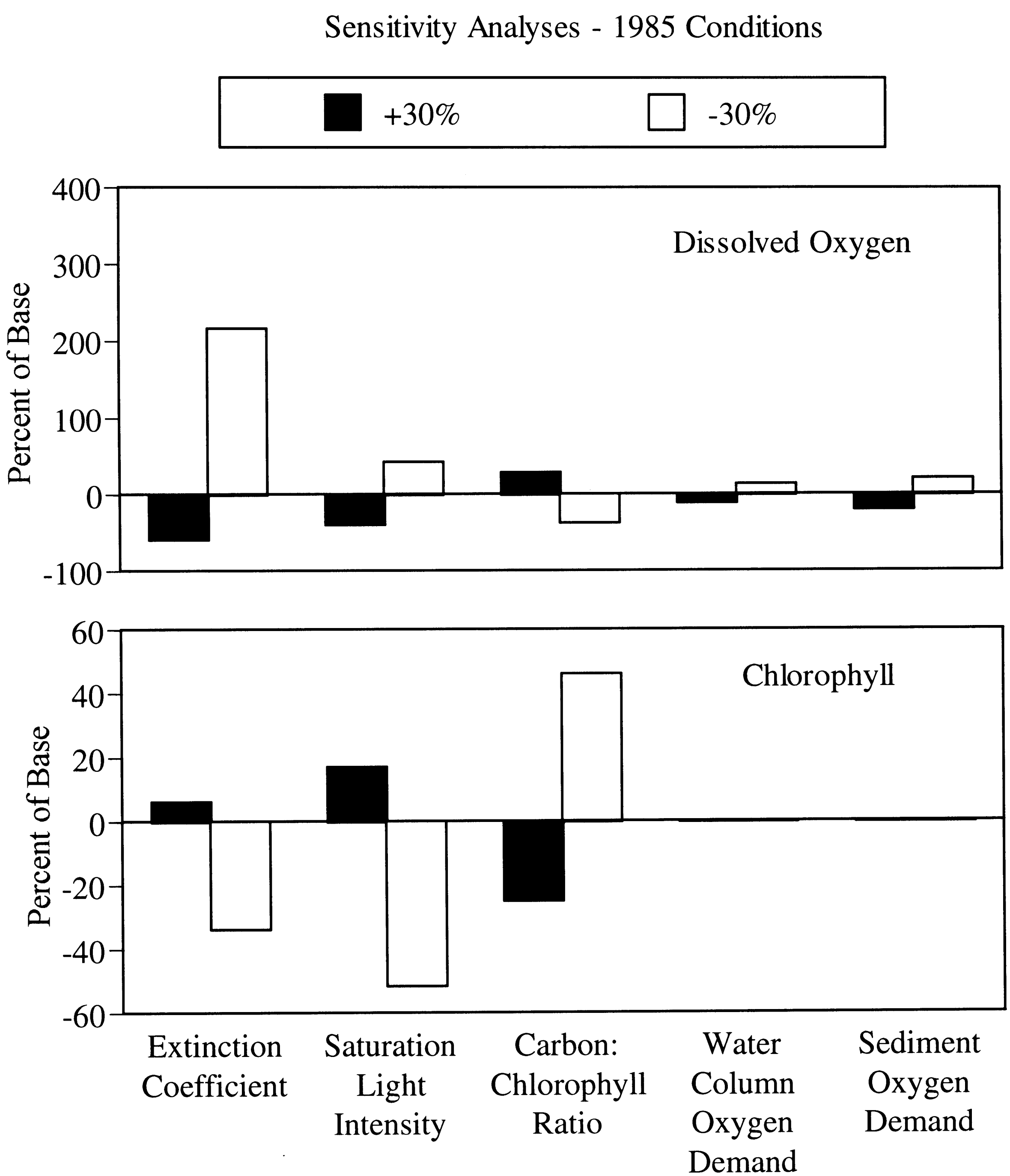


FIGURE 3.42. Sensitivity analyses for dissolved oxygen and chlorophyll concentrations in base calibration for 1985 conditions.

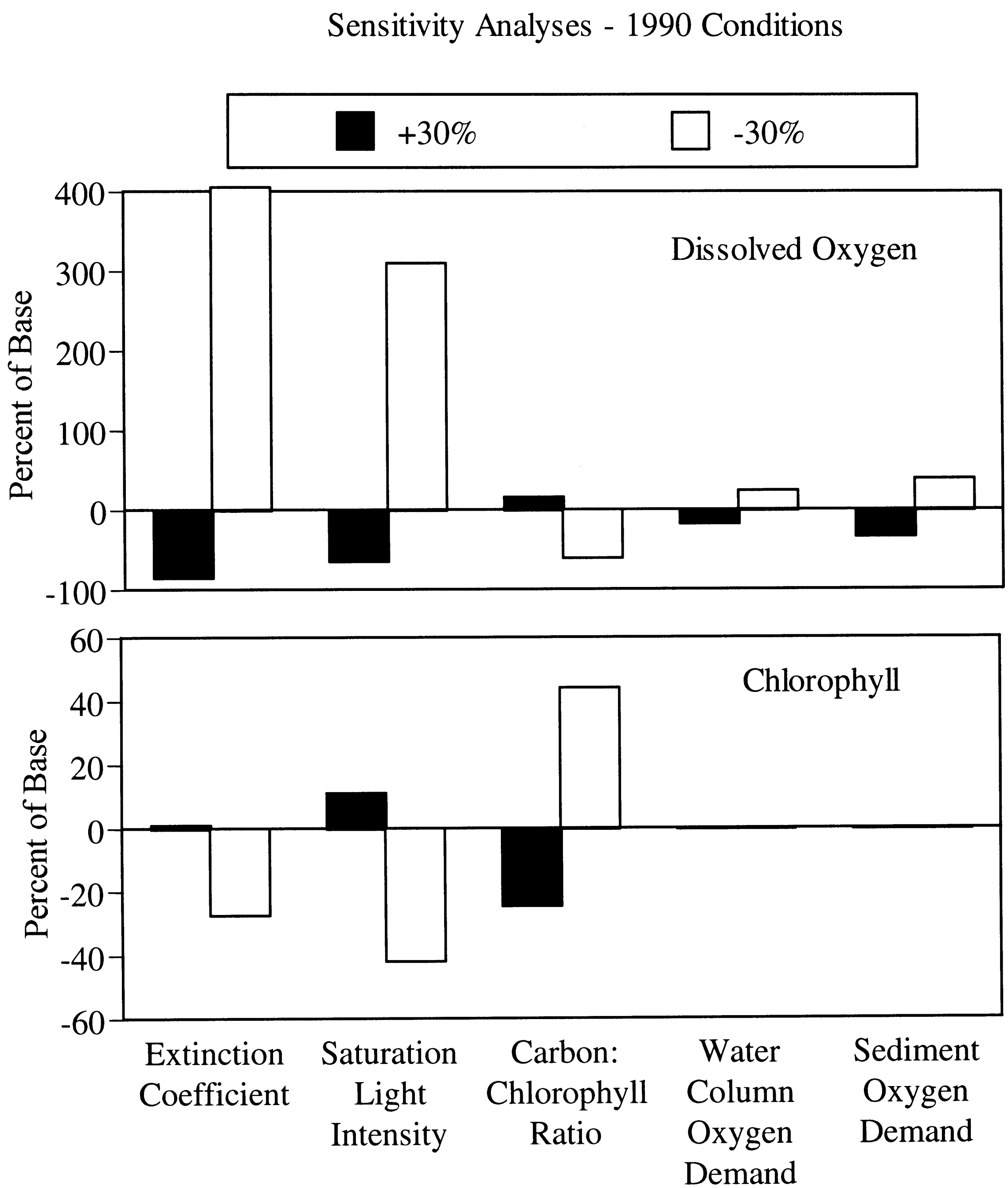


FIGURE 3.43. Sensitivity analyses for dissolved oxygen and chlorophyll concentrations in base calibration for 1990 conditions.

3.3.5 Discussion

3.3.5.1 FORECAST SIMULATIONS

The responses of chlorophyll and dissolved oxygen concentrations to reductions in nutrient loadings from the MAR are complex functions of internal model processes and external forcing functions. Part of this complexity is due to the fact that chlorophyll and dissolved oxygen are nonconservative, and are each tightly coupled to other state variables in the model (Figure 2.2). Chlorophyll is coupled to nutrient concentrations which, in turn, are coupled to MAR nutrient loadings and seaward and sediment nutrient boundary conditions. Chlorophyll is also directly coupled to chlorophyll concentrations at seaward boundaries. Finally, although chlorophyll is coupled to nutrient concentrations, phytoplankton growth rates in the model are strongly controlled by underwater light attenuation (Bierman et al. 1994a).

Dissolved oxygen is coupled to both intra-segment and seaward boundary concentrations of ammonium-N, phytoplankton carbon, and CBOD. It is also directly coupled to seaward and sediment dissolved oxygen boundary conditions. Dissolved oxygen concentration is much more strongly influenced than chlorophyll concentration is influenced by sediment boundary conditions. In the calibrated model for July 1990 conditions, SOD accounted for 22–30% of total oxygen-depletion rates in bottom waters (Bierman et al. 1994a). This is the principal reason why dissolved oxygen responses are more sensitive to differences in SOD than to differences in any other boundary conditions (Figure 3.36, top panel, and Figure 3.37). Chlorophyll responses are independent of differences in SOD or seaward boundary dissolved oxygen concentrations because chlorophyll concentration in the model is independent of dissolved oxygen concentration. Furthermore, differences in sediment nutrient fluxes have insignificant effects on surface-water chlorophyll concentrations because these internal loading sources are small relative to MAR mass loadings, and because sediment nutrient fluxes are prevented from directly affecting surface offshore segments due to vertical stratification of the water column (Bierman et al. 1994a).

Differences in responses of dissolved oxygen concentrations for 1990 conditions, as compared to 1985 and 1988 conditions (Figure 3.36, top panel), are probably due to specified differences in advective-flow magnitudes and directions (Figures 2.5–2.7). Much smaller flow magnitudes in 1990 may have affected the relative importance of physical transport versus chemical–biological processes for dissolved oxygen. Dissolved oxygen may have been affected more than chlorophyll because dissolved oxygen responses are more influenced by chemical–biological processes at the sediment–water boundary than are chlorophyll concentrations. This result is important because it indicates that estimates of water quality responses to changes in MAR nutrient loadings must be premised on specific assumptions for hydrometeorological conditions and advective flow fields on the Louisiana Inner Shelf (LIS).

There is considerable uncertainty in seaward boundary conditions for both the model calibration periods and the forecast simulations. The only calibration period for which field data were available outside the model's segmentation grid was July 1990 (Figure 2.3). Furthermore, even in July 1990 data were not available for all model state variables at each sampling station. For the July 1985 and August 1988 calibration periods, there were no water quality sampling stations outside the segmentation grid. For these years boundary concentrations were simply estimated from available data for the outermost field-sampling stations on cross-shelf transects.

Under nutrient-loading reductions from the MAR, it is reasonable to assume that seaward boundary conditions will change because these boundaries extend only to the 30–60-m bathymetric contours on the LIS. The assumption that these seaward boundary conditions will change by the same percentage as the corresponding nutrient-loading reduction was judged a reasonable way to bound the forecast simulations. A more realistic way to represent water quality on the LIS would be to extend the spatial domain of the model's segmentation grid so that its seaward boundaries are beyond the influence of MAR nutrient inputs. An obstacle to this approach is that additional field data would be required for determining far-field boundary concentrations, as well as for validating the model within the expanded segmentation grid.

Uncertainty also exists in SOD boundary conditions for both the model calibration periods and the forecast simulations. Boundary conditions for SOD in the calibrated model were based on aerobic benthic respiration measured using in-situ chambers (Rowe et al. 1992) and estimates of anaerobic metabolism (G.T. Rowe, Texas A & M University, personal communication, 1993). Although calibration values for SOD were based on field measurements, data were not available for all of the bottom offshore segments. In addition, measurements were available only for July 1990 and not for July 1985 or August 1988. Boundary conditions for SOD were assumed to be the same for all three model calibration periods.

Under nutrient-loading reductions from the MAR, it is reasonable to assume that SOD boundary conditions will change, because the primary source of SOD is net settling flux of particulate organic carbon originating from primary productivity in the water column. The assumption that SOD will eventually change by the same percentage as the corresponding nutrient loading was judged as being a reasonable way to bound the forecast simulations. A more realistic way to represent sediment–water oxygen fluxes would be to explicitly represent dissolved oxygen processes in the sediment and include an explicit dissolved oxygen mass balance between the water column and sediment segments (e.g., DiToro and Fitzpatrick 1993). This would require incorporating a separate sediment submodel and collecting additional field data for submodel validation.

All of the forecast results in this report represent ultimate steady-state responses. They provide no information on the time scales for potential water quality responses to MAR nutrient-loading reductions. On the basis of statistical correlations, Justić et al. (1993) estimated that there are time lags of approximately one and two months, respectively, between changes in MAR inflows and responses of surface net primary productivity and bottom-water dissolved oxygen concentrations on the LIS. Wiseman et al. (1997) observed good correlations between areas of mid-summer hypoxia and mean discharge for the preceding 11 months of August through June. They further indicated that, particularly for years with large floods followed by mild winters, the organic carbon substrate laid down by a major river flood may fuel the onset of hypoxia for more than a single year.

The ultimate response of bottom-water dissolved oxygen concentrations depends on the relative importance of SOD versus water-column processes in controlling bottom-water oxygen depletion rates. Total SOD consists of aerobic processes in the surficial sediments and anaerobic processes in deeper sediment layers. While surface-layer processes may respond to loading changes on seasonal-to-annual time scales, processes in deeper sediment layers may take many years to respond. For example, forecast results from a coupled water–sediment mass-balance model for Lake Erie (DiToro et al. 1987) indicated that the SOD component of total oxygen depletion rates did not reach steady-state until 5–10 years after changes in external nutrient loadings. Results from a similar coupled model for Chesapeake Bay (Cerco 1995a) showed that decade-long simulations were required to achieve a near-complete response to external loading reductions.

Although the differences in results between N and P loading reductions generally were not great (Figure 3.38), there was a tendency for responses to be somewhat greater for N than for P loading reductions. Turner and Rabalais (1991) suggested that N appears to be relatively more important than P in limiting primary productivity on the LIS. Lohrenz et al. (1997) found that primary productivity in shelf waters near the Mississippi Delta was significantly correlated with nitrate plus nitrite N concentrations and fluxes over a six-year period from 1988 to 1994. Fahnenstiel et al. (1995) reported that soluble N concentrations explained over 50% of the variability in phytoplankton growth rates in the northern Gulf of Mexico.

Differences in responses to N- versus P-loading reductions in the forecast simulations are a function of N:P ratios assigned to internal model parameters and external model-forcing functions in the model calibration. Marine phytoplankton are generally assumed to contain nitrogen and phosphorus in the Redfield ratio of N:P = 7.2 (by mass). Average TN:TP ratios in Mississippi and Atchafalaya River input loadings for the three calibration periods were 10.8 and 13.0, respectively. The average TN:TP ratio for seaward nutrient boundary conditions for the three calibration periods was 6.6. The Redfield ratio suggests that MAR nutrient inputs were slightly P-limiting and that seaward boundary nutrient concentrations were slightly N-limiting.

In the NECOP model, phytoplankton were assigned a stoichiometric N:P ratio of 5.0, based on calibration results to available phytoplankton and nutrient data. However, this stoichiometric ratio alone does not determine which nutrient is relatively more limiting in the model. Nutrient limitation in the model also depends on nutrient half-saturation constants and nutrient recycle processes. In any case, there were not large differences among the Redfield N:P ratio, the N:P stoichiometric ratio used in the calibrated model, or the N:P ratios in MAR nutrient loadings and seaward nutrient boundary concentrations (Justić et al. 1995). Consequently, the result that differences between N- and P-loading reductions were generally not great is consistent with assumptions underlying the model calibration.

Concerns have been raised (e.g., Turner and Rabalais 1991) that incompletely understood interactions among different nutrients (nitrogen, phosphorus, and silicon) and different phytoplankton functional groups (diatoms and nondiatoms) could result in major ecosystem changes. Rabalais et al. (1996) suggested that the combination of primary production changes and phytoplankton species shifts could affect subsequent carbon utilization, carbon flux pathways, and the areal extent or severity of hypoxia on the continental shelf. Officer and Ryther (1980) hypothesized that silicon limitation in highly eutrophic estuaries would cause shifts in phytoplankton species composition to nondiatom phytoplankton (which are not readily grazed), and would sink out of the euphotic zone and decompose on or near the seafloor, thereby causing large areas of hypoxia or anoxia. Dortch and Whitledge (1992) suggested an alternative hypothesis in which hypoxia on the northern Gulf of Mexico inner shelf would result from the decomposition of sinking large diatoms that are not completely grazed, especially in the spring when production is high (Lohrenz et al. 1990) and zooplankton biomass is low (Dagg and Whitledge 1991). Consequently, reduced silicon loads would decrease the dominance of large diatoms and the severity and extent of hypoxia. The essential point is that hypoxia is linked to the extent of primary production and to the fate of organic carbon from this primary production. In turn, the fate of this organic carbon is linked to the composition and abundance of phytoplankton species and to the grazing preferences of zooplankton. To address these questions, the model should include silicon as a potential limiting nutrient, diatom and nondiatom phytoplankton functional groups, and zooplankton as model state variables.

The reasons for differences in forecasted responses between dissolved oxygen and chlorophyll concentrations (Figure 3.39) are very complex. One reason is that the relative influences of MAR nutrient inputs, seaward boundary conditions, and bottom boundary conditions differ between the dissolved oxygen and chlorophyll state variables in the model. Another reason is that dissolved oxygen is coupled to more state variables in the model than chlorophyll. For example, dissolved oxygen responses represent the integrated effects of simultaneous changes, not only in dissolved oxygen processes per se, but also in CBOD, phytoplankton carbon (through endogenous respiration), and ammonium (through nitrification). Still another factor is that surface chlorophyll and bottom dissolved oxygen concentrations are coupled through the dependence of underwater light attenuation on phytoplankton self-shading. That is, reductions in surface-water chlorophyll concentrations can stimulate bottom-water primary productivity due to increased light penetration and, hence, increase bottom-water dissolved oxygen concentrations.

There was an overall tendency for forecasted responses of chlorophyll concentrations to be larger at greater distances from the Mississippi Delta (Figures 3.40 and 3.41). This is reasonable, because greater degrees of nutrient limitation would be expected to occur at greater distances from the principal input sources. However, the spatial dependence of dissolved oxygen concentration responses is more difficult to interpret. Results of diagnostic analyses with the calibrated model (Bierman et al. 1994a) suggest that primary productivity appears to be an important source of dissolved oxygen to bottom waters in the region of the Atchafalaya River discharge and farther west along the LIS. This region appears to be characterized by significantly different light attenuation–depth–primary productivity relationships than the area immediately west of the delta. Differences in forecasted spatial dependence of dissolved oxygen concentration responses for 1990 (Figure 3.41, top panel) compared with 1985 (Figure 3.40, top panel) and 1988 (not shown) were probably due in part to differences in hydrometeorological conditions and advective flow fields (Figures 2.5–2.7) among these three years.

3.3.5.2 SENSITIVITY ANALYSES

Dissolved oxygen and chlorophyll concentration responses in the model were very sensitive to variations in light-related parameters. This is consistent with observations that primary production in the northern Gulf of Mexico is very sensitive to changes in light intensity (Lohrenz et al. 1990, 1994). Model responses were not symmetric about equal plus-and-minus parameter variations because underwater light intensity decreases exponentially with depth, not linearly. These are important results because light-related parameters represent complex processes and are difficult to measure accurately in the environment.

There appears to be strong coupling between dissolved oxygen concentrations and primary productivity in bottom waters. This coupling is due to low underwater light attenuation and shallow water-column depths in the hypoxic region. Total suspended solid concentrations are relatively low (2–3 mg/L), and water-column depths range between only 16.1 and 30.3 m in the offshore portion of the model's spatial domain. Furthermore, water temperature and nutrient concentrations are relatively high during the summer stratified period, thus tending to magnify productivity responses.

The 1% depth is the compensation depth at which photosynthetic oxygen production approximately balances oxygen consumption due to phytoplankton respiration. Below Segments 11–14, west of the primary hypoxic region, the 1% depth estimated from light-extinction coefficients in the calibrated model is greater than the total depth of the water column. This implies that phytoplankton are a net source of dissolved oxygen in this region. This model result is consistent with independent observations confirming that considerable primary production occurs on the sediment surface in the model's spatial domain, especially further west along the inner shelf. For 12 locations at which measurements were conducted, G.T. Rowe (Hendee 1994) has reported that bottom productivity rates averaged approximately 30% of water-column productivity rates.

Responses for chlorophyll concentrations in the model appear counterintuitive because they increase in response to increased light attenuation and saturation light intensity (+30% changes) and decrease in response to decreased light attenuation and saturation light intensity (–30% changes) (Figures 3.42 and 3.43). Although this behavior could be an artifact of the model, a possible reason could be photoinhibition in surface waters. Phytoplankton growth rate in the model increases as a function of increasing light intensity up to a saturating level and then decreases (photoinhibition) with further increases in light intensity (Thomann and Mueller 1987). For model surface segments, the computed light-saturation depth is approximately 5–7 m. This implies that phytoplankton growth rates in the model are limited by photoinhibition in nearshore waters (Segments 1–7) and in major portions of surface offshore waters (Segments 8–14). Increases in light attenuation and saturation light intensity stimulated surface primary productivity in the model because they reduced photoinhibition in the surface segments.

Chlorophyll concentration responses to variations in the carbon:chlorophyll (C:C) ratio depend on P:C and N:C stoichiometric ratios in the phytoplankton. An increase in the C:C ratio decreases the amount of chlorophyll that can be produced for given P:C and N:C ratios, assuming no changes in nutrient loadings. The opposite is true for a decrease in the C:C ratio.

Dissolved oxygen responses in bottom waters are confounded because phytoplankton in the model are represented as carbon, and chlorophyll is only a display parameter. However, the self-shading term in the submodel for underwater light attenuation is a function of computed chlorophyll concentration. Consequently, when the C:C ratio is increased, chlorophyll concentrations decrease, thus causing increases both in underwater light intensity in the water column and in dissolved oxygen concentrations.

Dissolved oxygen responses in the model were sensitive to variations in both SOD and water-column oxygen demand (the CBOD decay rate). Average dissolved oxygen responses between 1985 and 1990 conditions were approximately linear with variations in SOD. Responses to variations in SOD were greater than responses to variations in the CBOD decay rate. This reinforces the importance of incorporating a separate submodel to explicitly represent dissolved oxygen processes in the sediment.

Dortch et al. (1992) hypothesized that the high productivity observed on the LIS is maintained by nitrogen recycling within the water column, thus greatly amplifying the effect of high riverine nitrate inputs. Results of numerical experiments in which water-column nutrient remineralization rates were “turned off” (Bierman et al. 1994a) indicated that chlorophyll concentrations decreased substantially in almost all model spatial segments. Dissolved oxygen concentrations in bottom waters responded not only to decreases in chlorophyll concentrations (due to decreases in primary productivity), but also to decreases in ammonium concentrations (resulting from decreases in nitrification). These results illustrate the complex interactions among phytoplankton, nutrient, and dissolved oxygen dynamics on the LIS.

It is difficult to investigate uncertainties due only to physical transport processes in the model, because the influence of these processes is strongly coupled to specified values for seaward boundary conditions. Some of the uncertainties due to physical processes are reflected in the different model results among 1985, 1988, and 1990 conditions. Bierman et al. (1994b) presented results for sensitivity of model responses to changes in dispersive mixing across seaward boundaries and vertical dispersion. The principal cross-shelf transport component in the model was dispersive mixing because, with the exception of some off-shelf advective flow in 1990 (Figure 2.7), there were no cross-shelf advective flows specified in the model applications. Bottom-water dissolved oxygen concentrations were found to be approximately proportional to changes in cross-shelf dispersive mixing and relatively insensitive to changes in vertical mixing. Corresponding chlorophyll concentration responses were relatively insensitive to both of these changes. These results reinforce the need for a better quantitative understanding of physical oceanographic processes in the northern Gulf of Mexico.

CHAPTER 4

Recommendations

4.1 MISSISSIPPI RIVER BASIN

4.1.1 Monitoring

Data such as those generated by Metropolitan Council Environmental Services and by routine U.S. Geological Survey (USGS) monitoring efforts are a vital component of our ability to make statements (1) about the current state of the system and (2) about the potential applicability of current or future modeling frameworks. Therefore, it is critical that these monitoring programs be continued.

Additional monitoring sites need to be established in the main channel of the Upper Mississippi River to be able to evaluate the extent of nutrient retention and loss within the lock-and-dam system. A monitoring program needs to be established to clarify the extent of nutrient retention in the Lower Mississippi: Is it a delivery pipe, or are significant input, output, and cycling of nutrients occurring?

Better and more water quality monitoring also needs to be conducted at finer spatial scales to establish effects of changes in land-management practices on resulting water quality and nutrient concentrations and loads, and also to evaluate scale effects on nutrient retention. For these purposes, it would be beneficial to establish a system for long-term monitoring and research at the scale of fields and minor watersheds similar to the long-term ecological research (LTER) sites funded by the National Science Foundation. A broad, interdisciplinary approach that includes landscape, aquatic, and socioeconomic measurements should be taken in the research and monitoring done at such intensively studied sites so that we can improve our understanding of the critical processes occurring at the watershed and drainage basin scales. Finally, monitoring of patterns of fertilizer use is critical for targeting improvements in management practices within the previously identified problem areas.

4.1.2 Research

Although agricultural scientists have been developing and studying improved management practices to minimize off-site impacts of agricultural production for many years, much more research is needed, especially at the whole-farm, watershed, and larger scales. There is a great need for research on the impacts of large-scale, confined animal-feeding operations and for studies on ways to minimize those impacts.

Nutrient cycling in riverine systems needs further quantification. We need much better information concerning rates of all major nitrogen-cycle processes, but especially nitrification and denitrification and the factors affecting these processes in the Upper and Lower Mississippi River, the larger tributaries, and the smaller, nitrate-rich streams that drain agricultural lands. Similarly, information is lacking on the mechanisms of phosphorus retention and the factors affecting those processes.

In many systems, shifts toward dominance of the river plankton by species of blue-green algae (cyanobacteria) may occur in eutrophic or hypereutrophic rivers. It is important to assess whether the mechanism(s) causing these shifts are the same as, or different from, those causing cyanobacterial blooms in lakes and reservoirs. Results from a study of the freshwater portion of the Potomac suggest a possible effect of TN:TP ratios on cyanobacterial blooms in rivers that parallels the effects observed in freshwater lakes (Limno-Tech 1991; V.H. Smith 1998). These algal cells, and the nutrients contained within them, can form the inoculum for nuisance cyanobacterial blooms in downstream reservoirs and estuarine bays that have much longer hydraulic residence times than the upstream river channels.

As in lakes and reservoirs, growth limitation by N or P availability can occur in rivers, and standing crops of both benthic and suspended algae in general tend to increase with eutrophication (e.g., Moss et al. 1989; Dodds et al. 1997, 1998; Ibelings et al. 1998; Van Niewenhuyse and Jones 1996). However, the intensity and frequency of occurrence of nutrient limitation have been much better studied for benthic algae than for suspended algae (Köhler and Gelbrecht 1998). Careful assessments of N versus P limitation (and of nutrient versus light limitation) should be performed in a wide variety of rivers and streams using algal enrichment bioassays, algal tissue nutrient analysis, and measurements of inorganic N and P to better characterize the intensity, spatial extent, and seasonality of algal nutrient limitation in flowing waters. In addition, critical assessments should be made of the importance of light versus nutrient limitation of algal growth in streams and rivers. These assessments should build upon and complement the parallel studies done in lakes by V.H. Smith (1982), Canfield and Bachmann (1981), and Hoyer and Jones (1983).

Dodds et al. (1998) recently proposed new trophic limits and critical nutrient concentrations for flowing waters that parallel the categories that have been defined for lakes and reservoirs. This approach should be continued and refined to accelerate the shift from subjective, potentially biased indices of water quality to more quantitative and objective criteria for eutrophication assessment in streams and rivers.

4.1.3 Modeling

It is important that we encourage the further development of regression-based models that relate nutrient-related water quality variables to stream trophic state and nutrient loading from the watershed. The semi-empirical approach of Vollenweider (1969, 1975) and others has proven to be useful worldwide in managing eutrophication in lakes and reservoirs, and this approach should be encouraged for flowing waters as well. Recent comparative studies (e.g., Dodds et al. 1997, 1998; Van Niewenhuyse and Jones 1996) suggest that this approach can be extended successfully to the nutrient management of streams and rivers. It is critical to develop Vollenweider-type indices for N, as well as for P. In addition, the N:P ratio hypothesis needs further clarification. The semi-empirical models for algal growth should focus not only on response variables, such as the mean biomass benthic or suspended algae, but also on the peak biomass that can be attained during the year or growing season.

In addition to developing models relating important variables of concern (e.g., algal biomass) to in-stream nutrient concentrations, it is vitally important to develop further quantitative models that in turn relate watershed-based nutrient loading to stream-water nutrient concentrations and transport in the MRB. These models can be developed at different levels of complexity, scale, and user friendliness, ranging from spreadsheet models (Dodds et al. 1997), to the SPARROW model (R.A. Smith et al. 1997), to more complex models such as BATHTUB, QUAL2E, and WASP5. The development of such models is essential to provide quantitative links between land use, improved land management, and in-stream water quality. In addition, these models will help managers assess the critical question of response time (e.g., How rapidly will the water quality of a given stream or hydrologic cataloging unit improve following a 10% reduction in fertilizer application rates?).

No satisfactory framework currently exists to predict the quantitative effects of changes in stream nutrient concentrations on either fish yield or fish species composition. Thus, no quantitative statements can be made about the potential effects of nutrient-loading restrictions on fisheries throughout the MRB. Attempts should be made to develop such models.

4.2 GULF OF MEXICO

4.2.1 Monitoring

The principal obstacle to reducing uncertainties in quantifying linkages between Mississippi–Atchafalaya River (MAR) nutrient loads and water quality responses in the northern Gulf of Mexico is the lack of a sufficiently comprehensive database. Although the existing database is comprehensive in many respects, the data were acquired primarily to characterize water quality responses, not to provide data quantifying load-response relationships or principal controlling processes. To pursue these objectives, a quantitative conceptual model of ecosystem structure and function in the northern Gulf of Mexico should be conceived and then used as a foundation for developing future monitoring plans.

Monitoring should include a nested hierarchy of spatial scales, temporal scales, and measured parameters. It should be integrated with remote-sensing observations, as well as with quantitative models for physical, chemical, and biological processes. Measurements for a limited number of parameters should be conducted at high frequency at a small number of strategic spatial locations. Measurements for a comprehensive suite of parameters should be conducted at a lower frequency on a shelfwide scale, and at intermediate scales for selected parameters and in-situ process rates and fluxes.

There is a basic need for good physical oceanographic data on water movements and other physical processes. Long-term, continuous moorings should be maintained at strategic locations to measure current speed and direction, conductivity, salinity, temperature, and dissolved oxygen at multiple depths in the water column. Synoptic cruises should be conducted on approximately a monthly basis during periods of seasonal stratification. These cruises should include measurements for a comprehensive suite of water quality parameters. The number of stations should be sufficient to characterize (1) spatial variability within the hypoxic region out to approximately the 30–60 m depth contours and (2) seaward boundary conditions out to approximately the 100–200 m depth contours and deeper near the Mississippi Delta.

It is important to measure attenuation coefficients for downwelling irradiance and other correlated parameters. These parameters should include chlorophyll, inorganic and organic suspended solid concentrations, Secchi disk depth, beam-attenuation coefficient, and salinity. It is not sufficient to measure only underwater light-attenuation coefficients and specify these values directly to a mass-balance model. If this were done, the model would only be valid for the particular conditions under which the attenuation coefficients were measured. Such a model would not be valid for conducting any sensitivity analyses or forecast simulations that involved changes in chlorophyll concentrations or any other parameters that co-vary with light-attenuation coefficients. A credible and useful mass-balance model requires an independent submodel for underwater light attenuation as a function of chlorophyll concentrations and any other significantly correlated parameters.

The strong coupling between dissolved oxygen concentrations, primary production, and changes in algal communities in bottom waters emphasizes the importance of in-situ measurements of primary production, in conjunction with measurements of light-attenuation coefficients. These measurements should include both planktonic and benthic algal types and their production rates.

Comprehensive field data are required for specifying external model-forcing functions, as well as for comparing model outputs. The most important of these forcing functions are nutrient loadings from the MAR. Monitoring of these loadings should include both systematic and high-flow event sampling. Systematic sampling should be conducted on at least a monthly basis, and high-flow events should be sampled perhaps 20–30 times annually. Other important forcing functions that should be measured include atmospheric deposition, incident solar radiation, sediment–water nutrient fluxes, and sediment oxygen demand.

4.2.2 Research

Field data generated by a comprehensive monitoring program are necessary but not sufficient for developing, calibrating, and validating quantitative water quality models. There is not yet a complete understanding of the physical, chemical, and biological processes that influence water quality responses in the northern Gulf of Mexico to changes in MAR nutrient loadings. Research is needed to better understand these processes and to provide information for representing and parameterizing them in quantitative models.

Important unresolved research questions include the following:

- Relationships among saturation light intensities, underwater light attenuation, and specific growth rates for indigenous species.
- Factors controlling benthic primary productivity.
- Factors controlling oxygen-depletion rates in water columns.
- Factors controlling sediment–water nutrient fluxes and sediment oxygen demand.
- Magnitudes and seasonal variability of particulate organic carbon (POC) and particulate organic nitrogen (PON) settling fluxes in the water column.
- Importance of shifts in phytoplankton species in influencing fate pathways for organic carbon.
- Importance of silica limitation and its role in influencing shifts in phytoplankton species.
- Influence of phytoplankton–zooplankton interactions on fate pathways for organic carbon.
- Rate and extent of nutrient remineralization in the water column.
- Importance of nitrification and denitrification in sediments on the total nitrogen budget for the northern Gulf of Mexico.
- Relative importance of atmospheric loadings to the northern Gulf of Mexico.
- Usefulness of satellite imagery to track plumes and plume dynamics, and the Louisiana Coastal Current to gain understanding of phytoplankton distributions in surface layers.

4.2.3 Modeling

To more accurately represent the physical, chemical, and biological processes controlling water quality responses in the northern Gulf of Mexico to changes in MAR nutrient loadings, future modeling work should include the following:

- Advective flows and dispersive mixing coefficients in the water quality model should be determined using the output of a hydrodynamic model of Gulf of Mexico circulation.
- The temporal domain of the present Nutrient Enhanced Coastal Ocean Productivity (NECOP) model should be extended to include a continuous, time-variable representation of water quality conditions over the complete annual cycle.
- The spatial domain of the present model segmentation grid should be extended so that its seaward boundaries are beyond the influence of freshwater and nutrient inputs from the MAR.
- The vertical resolution of the present model segmentation grid should be refined to better represent near-bottom hypoxia on the Louisiana Inner Shelf (LIS).
- The horizontal spatial resolution of the present model should be sufficiently refined to assess changes in the area and volume of hypoxia under different management strategies.
- Finer spatial–temporal resolution should be employed to represent the dynamics of nearshore waters (shore to 60 m depth) and linkages with estuaries and offshore waters.
- Model calibration and validation should be conducted over several years (three or more) with different MAR inflows (average, wet, and dry years).
- The conceptual framework of the model should be expanded to include a sediment diagenesis submodel and explicit representation of nutrient and dissolved oxygen mass balances between water column and sediment segments.
- The conceptual framework of the model should be expanded to include all principal phytoplankton functional groups, including diatoms, and silica as a potential limiting nutrient.
- The water quality model should include a separate submodel for underwater light attenuation as a function of background color, biotic solids (phytoplankton), and abiotic solids.

These modeling needs cannot be met independently of the monitoring and research needs presented above. Models are only tools for synthesizing environmental data and cannot be used as substitutes for these data. For a model to be useful it must have the capabilities for addressing the principal management questions. For a model to be scientifically credible, there must be adequate field data for specification of external forcing functions and for comparison with model output. In summary, there must be compatibility among the management questions, the model capabilities, and the available field data.

CHAPTER 5

Conclusions

5.1 MISSISSIPPI RIVER BASIN

The vast Mississippi River Basin encompasses landscapes with highly diverse physical, geological, and biological characteristics, including temperate and alpine forests; rich agricultural croplands; prairie and pastureland; arid lands; and swamps, bogs, bayous, and other wetlands. Water chemistry and water quality conditions in the numerous small and large tributaries that drain into the main channels of the MRB reflect this variety of landscapes and are similarly diverse. Under these circumstances, it is difficult to make general statements about the effects of nutrient source reductions on water quality in the MRB. Instead, the following conclusions focus on those parts of the MRB most responsible for the high nutrient loads to the Gulf of Mexico, especially the agriculturally dominated Midwest and North Central states (i.e., the Corn Belt). Rivers and streams in this region typically have high nutrient (N and P) concentrations and often have high levels of suspended solids. Because of the resulting low water clarity, they tend to have limited growths of submerged macrophytes. Instead, primary producers in these systems are dominated by phytoplankton. Small and mid-sized streams in this region also tend to be altered hydrologically as a result of wetland drainage, channelization, and development of dams.

Further reductions in point-source contributors of nutrients to rivers and streams in the MRB will have commensurate effects on nutrient concentrations and loads immediately downstream, but the intensity of these effects will be less as the water flows further downstream because of dilution (from other tributaries) and in-stream processes of nutrient loss and retention. Because point sources account for a small proportion (~10%) of the total nutrient load of the Mississippi River at its mouth, further controls on point sources will have a relatively small effect on total MRB loads of nutrients to the Gulf of Mexico, but such controls may have significant effects on nutrient loads and concentrations in local areas. For example, phosphorus loads from wastewater treatment plants in the Twin Cities of Minneapolis and St. Paul, MN, contribute more than 50% of the total phosphorus input to Lake Pepin, a natural impoundment of the Mississippi River about 40 miles south of the Twin Cities area. Nonetheless, even if these point sources were completely eliminated, nonpoint sources of nutrients—primarily from agricultural landscapes in the MRB—that flow into the Mississippi River at the Twin Cities would be sufficient to maintain Lake Pepin in its moderately eutrophic condition.

Agricultural lands are the largest contributors of nonpoint-source nutrients to rivers and streams in the MRB, especially in the agriculturally dominated Midwest and North Central states. The processes by which nutrients are transported through agricultural lands and delivered to receiving waters differ for nitrogen and phosphorus. The former is associated more with subsurface drainage, and artificial (tile) drainage is an important factor contributing to high N losses. The latter is associated more with surface runoff, and soil erosion is an important factor contributing to high P losses. Consequently, options for reducing losses also must be considered separately.

Nitrate loading to surface waters from cultivated land generally is a result of one or more of the following: heavy precipitation or snowmelt; subsurface drainage systems; soils with high organic content; and applications of N fertilizer or manure in excess of agronomic recommendations.

Significant reductions in nitrate losses to surface waters can be achieved by a wide variety of improved management practices (IMPs), such as:

- increasing the lateral spacing of subsurface tile drainage to 20 m rather than 5–10 m;
- controlling water-table levels to promote denitrification within the soil column;
- routing tile-drainage effluent through wetlands, grass buffer strips, or riparian forest buffers;
- changing from row-cropping to perennial-cropping systems;
- planting a cover crop of rye grass during the fall and the winter;
- switching from conventional to ridge-tillage or other reduced tillage practices;
- switching from fall to spring application of fertilizer; and
- limiting the application of nitrogen fertilizer and manure to agronomically recommended rates.

For some of the above IMPs, reductions in losses are on the order of 10–20% of baseline conditions, but others have been shown to reduce losses by up to 90% in specific field studies. From a practical standpoint, not all of the above options are equally viable. Moreover, for significant reductions in nitrate loads to occur at the scale of watersheds and basins, there must be widespread adoption of improved nutrient management practices.

An early version of a large-scale simulation model being developed for the entire U.S. predicted that nitrogen fertilizer applications in the Corn Belt states could be reduced significantly without seriously reducing the national production of these crops (in part because of a compensating increase in crop production that most likely would result outside the MRB). The model also predicted that N discharged from croplands in the Corn Belt would decrease by only a small amount if this strategy were adapted by or imposed on all farmers in the Corn Belt, and that P discharged from croplands would not change significantly by a program focusing only on reducing fertilizer applications to crops. Because the model is still under development and the current version does not simulate some potentially important conditions (in particular, the effects of tile drainage on nutrient export from croplands), these early conclusions should not be used as the basis for policy and management decisions. The model should be further developed and tested in the field.

Very few directly measured values of nutrient loss or retention exist for flowing waters in the MRB, and existing monitoring data are not adequate to address this important issue. Nutrient retention and losses estimated by a simple first-order model for in the streams and rivers in the Upper and Lower basins of the MRB were: (1) ~35–40% for total nitrogen (TN) in small tributaries, and ~20% in mainstem rivers of these regions; and (2) ~28–37% for total phosphorus (TP) in small tributaries and negligible in the mainstem channels.

Violations of numerical water quality standards for dissolved oxygen, pH, nitrate, and un-ionized ammonia are uncommon in MRB rivers and streams under current and recent conditions. It should be noted, however, that at present there are no numerical standards for nutrients in water bodies relative to their potential to cause eutrophication problems (e.g., the nitrate standard applies to drinking water). Current levels of CBOD and NBOD do not appear to be high enough to cause problems of low dissolved oxygen, except perhaps in isolated situations within MRB rivers and streams.

Nonetheless, most states in the MRB have substantial numbers of river miles that suffer use impairment related to nutrient conditions or do not completely fully support three important resource uses—aquatic life support, fish consumption, and swimming. Reductions in nutrient concentrations, if significant, have the potential to improve this situation.

In an effort to quantify the extent of water quality impairment in MRB rivers and streams, we proposed eutrophication criteria for TP and TN, based on recent research on such criteria in flowing waters. About 30–55% of the hydrologic cataloging units (HCUs) of the Ohio, Lower Mississippi, and Tennessee subbasins exceed the proposed eutrophic criterion for TP concentration in flowing waters, and 16–40% of the HCUs in these three regions exceed the proposed flowing-water criterion concentration for TN. Higher exceedance frequencies occur in the Upper Mississippi, Arkansas-Red, and Missouri sub-basins: ~80% of the HCUs exceed the TP trophic criterion, and 70–75% exceed the TN criterion. A 30% reduction in TP concentrations is required in the Upper Mississippi. Arkansas–Red, and Missouri regions to obtain a 10% reduction in the number of HCUs that exceed the stream trophic criterion for TP. A 15% reduction is required in the Lower Mississippi, Tennessee, and Ohio regions to achieve a 10% reduction in the rate of exceedance.

Under current conditions, both phosphorus and nitrogen act as the nutrients limiting plant growth in various flowing waters of the MRB. Analysis of N:P ratios for such flowing waters across the MRB indicates that 69% of the waters fall into the combined N+P and P-limited class, and 31% of the sites exhibited potential N limitation.

Although the quantitative analysis of nutrient loading/trophic state response relationships in flowing waters is in its infancy (compared with the situation for eutrophication in lakes), an empirical relationship is available to predict the improvements in chlorophyll that would occur on average following reductions in TP concentrations in river reaches of fixed catchment size. The relationship predicts an 18% reduction in chlorophyll if stream-water TP is reduced from 125 to 100 µg/L, and a 52% reduction in chlorophyll is predicted for a 25 µg/L decrease in TP if the initial stream-water TP is 50 µg/L. The above relationship and calculations lend support to the supposition that efforts to reduce nutrient levels in flowing waters of the MRB will lead to lower concentrations of river phytoplankton and also to somewhat greater water clarity in downstream waters. However, considerable scatter exists in chlorophyll–TP correlations, and many factors besides TP concentrations influence the production of algae in a river or lake.

If reductions in N and P levels increase underwater light, submersed aquatic macrophyte distribution will expand in the Upper Mississippi River. The effects on water quality will be beneficial, both locally and system-wide. Increased macrophyte abundance may augment nutrient retention significantly, leading to lower delivery rates of nutrients to the Gulf of Mexico than predicted from direct effects of external source reductions.

Because rivers and streams in the MRB most likely will remain moderately productive systems, sport fisheries probably would not be affected strongly by changes in nutrient inputs from improved land management practices and additional controls on point sources. Any declines that might occur in total biomass production most likely would be more than compensated for by habitat and other improvements that would promote game fish over rough fish populations.

5.2 GULF OF MEXICO

The results presented in this report are from an ongoing research program and should be considered preliminary and provisional in nature. Dissolved oxygen and chlorophyll concentrations on the Louisiana Inner Shelf (LIS) appear responsive to reductions in N and P loadings from the Mississippi–Atchafalaya River (MAR). For a given reduction in MAR N or P loadings, there are large uncertainties in the magnitudes of dissolved oxygen and chlorophyll concentration responses. These uncertainties are the result of four principal factors:

- lack of information on relationships between MAR nutrient loadings and seaward boundary conditions;
- lack of information on relationships among light attenuation, water-column depth, and primary productivity;
- lack of information on relationships between MAR nutrient loadings and sediment oxygen demand; and
- variability in hydrometeorological conditions in the northern Gulf of Mexico.

In response to nitrogen-loading reductions of 20–30%, average dissolved oxygen concentrations increased by less than 5% under constant boundary conditions and by 25–90% under reduced boundary conditions, depending on differences in hydrometeorology. The range of these increases was 15–50% when results for constant and reduced boundary conditions were averaged.

In response to nitrogen loading reductions of 20–30%, average chlorophyll concentrations decreased by less than 2% under constant boundary conditions and by less than 15% under reduced boundary conditions, depending on differences in hydrometeorology. These decreases ranged between 5% and 10% when results for constant and reduced boundary conditions were averaged.

Although differences in results between N- and P-loading reductions generally were not large, there was a tendency for responses to be somewhat greater for N-loading reductions than P-loading reductions, especially for dissolved oxygen under reduced boundary conditions.

There was no evidence of significant interactions between N and P loading reductions. Results of simulations in which N and P loadings were reduced simultaneously were generally consistent with results of simulations in which the more limiting of the two nutrients was reduced by itself.

In general, average dissolved oxygen concentrations were more responsive than average chlorophyll concentrations with increasing distance from the Mississippi Delta. Magnitudes of these response differences depended on differences in hydrometeorological conditions.

The spatial distribution of responses for average dissolved oxygen concentrations was highly dependent on differences in hydrometeorological conditions. In general, responses for average chlorophyll concentrations tended to increase with increasing distance from the Mississippi Delta.

Dissolved oxygen concentrations were very sensitive to variations in underwater light-extinction coefficients and saturation light intensities, relative to base calibration results. Model responses were not symmetric about equal plus-and-minus variations in model parameters.

Dissolved oxygen concentrations were sensitive to variations in both sediment oxygen demand (SOD) and water-column oxygen demand (the CBOD decay rate). Average responses between 1985 and 1990 conditions were approximately linear with variations in SOD. Responses to variations in SOD were greater than responses to variations in the CBOD decay rate.

Dissolved oxygen concentrations were sensitive to variations in cross-shelf dispersive mixing and relatively insensitive to changes in vertical mixing. Average responses were approximately proportional to changes in cross-shelf mixing.

Chlorophyll concentrations were sensitive to variations in underwater light-extinction coefficients, saturation light intensities, and carbon:chlorophyll ratios, relative to base calibration results, and model responses were not symmetric about equal plus-and-minus variations in model parameters. Chlorophyll concentrations were relatively insensitive to variations in cross-shelf dispersive mixing and vertical mixing.

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