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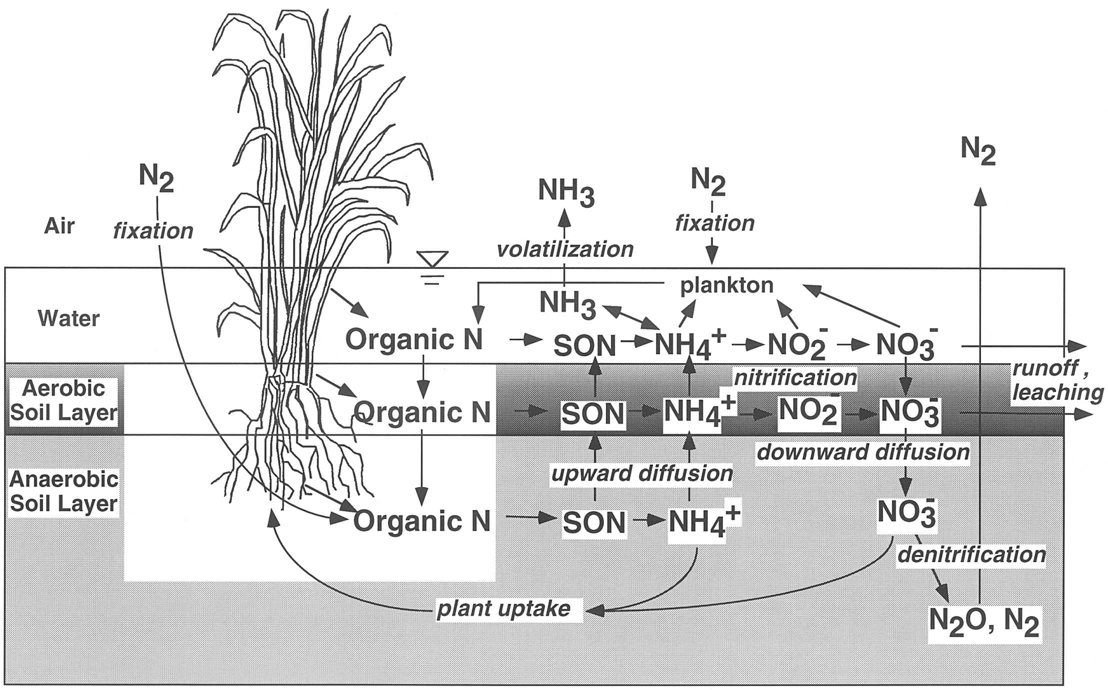
NOAA COASTAL OCEAN PROGRAM
Decision Analysis Series No. 19

Reducing Nutrient Loads, Especially Nitrate–Nitrogen, to Surface Water, Ground Water, and the Gulf of Mexico

Topic 5 Report for the Integrated Assessment on Hypoxia in the Gulf of Mexico

William J. Mitsch, John W. Day, Jr.,
J. Wendell Gilliam, Peter M. Groffman, Donald L. Hey,
Gyles W. Randall, and Naiming Wang

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 fifth 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: Nitrogen transformation in wetlands.

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May 1999

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List of Abbreviations and Acronyms

BACI	Before-After, Control-Impact Analysis
BOD	Biochemical Oxygen Demand
CAAA	Clean Air Act Amendments
CNG	Compressed natural gas
CP	Chisel plow
CRP	Conservation Reserve Program
CT	Conventional tillage
CZARA	Coastal Zone Act Reauthorization Amendments
DIN	Dissolved inorganic nitrogen
Eff	Efficiency
EGR	Exhaust gas recirculation
ET	Evapotranspiration
EW	Experimental wetland
HDE	Heavy-duty engine
HEGO	Heated exhaust gas oxygen sensors
HLR	Hydraulic loading rate
IHNC	Inner Harbor Navigation Canal
LDT	Light-duty truck
LDV	Light-duty vehicle
LEV	Low-emission vehicle
LNB	Low-NO_x burner
LR	Loading rate
MP	Moldboard plow
MRB	Mississippi River Basin
N	Nitrogen
NI	Nitrapyrin
NO_x	NO and NO_2 (gas)
$NO_2 + NO_3$	Nitrite plus nitrate (dissolved in water)
NO_3	Nitrate
NPS	Nonpoint source
NRC	National Research Council
NT	No tillage
OSDS	On-site sewage disposal
RR	Removal rate
RSN	Residual soil nitrate
RT	Ridge tillage
SCR	Selective catalytic reduction
SNCR	Selective noncatalytic reduction
TKN	Total Kjeldahl nitrogen
TN	Total nitrogen
TP	Total phosphorus
TSS	Total suspended solids
UEGO	Universal exhaust gas oxygen sensors
ULEV	Ultra-low-emission vehicle
USDA	United States Department of Agriculture
USEPA	United States Environmental Protection Agency
ZEV	Zero-emission vehicle

Acknowledgments

We would like to thank Terry LeMaster for her cheerful assistance in arranging all of the committee meetings, both in St. Louis and over the phone. Eric Domyan provided a valuable literature review on point-source technology. Sarah Harter and Virginie Bouchard assisted with many aspects of this study. Raghavan Srinivasan of Texas A&M participated in our first meeting's discussion. Ruthmarie H. Mitsch assisted with manuscript proofing.

We appreciate the efforts of Don Scavia, Mike Dowgiallo, and others at NOAA's Coastal Ocean Program in Silver Spring, MD, for their leadership and assistance in the project. We also appreciate the support provided by the Gulf of Mexico Program of the U.S. Environmental Protection Agency, Stennis Space Center, Mississippi, on Grant #MX984357-98-0.

Jerry Hatfield coordinated a comprehensive and useful peer-review process. We would also like to thank the six anonymous reviewers for the time that they took to do provide very useful comments to the authors.

Riparian information on habitat was largely taken from USDA's *Riparian Forest Buffer Handbook for the Chesapeake Bay Watershed*, 1997. Thanks to Al Todd for letting us use the draft version of this document. Don Goolsby was very accommodating in providing much-needed data on nitrogen sources.

Thanks go to our home institutions for allowing us the time to complete this project: The Ohio State University, Louisiana State University, University of Minnesota, North Carolina State University, Institute of Ecosystem Studies, and The Wetlands Initiative.

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 goal of this report to identify and evaluate approaches for solving the problem of the hypoxia in the Gulf of Mexico. This zone of low dissolved oxygen, which covers an area from 13,000 to 20,000 km^2 off the shore of Louisiana, has been shown to be due to excess nutrients, particularly nitrate–nitrogen, being transported to the Gulf from the Mississippi River Basin. To accomplish our goal, we (1) reviewed appropriate literature on methods for controlling nutrients, particularly nitrate–nitrogen, from entering waterways; (2) evaluated these methods to reduce the amount of nitrogen released to streams and rivers; (3) put the methods in the context of the entire Mississippi River Basin and the significance of the sources; and (4) presented recommendations for the most reasonable combination of approaches that would be necessary to solve the problem.

Techniques reviewed included on-farm practices, created and restored wetlands and riparian zones, controlled drainage systems, stormwater runoff control, atmospheric controls on mobile and stationary sources, point-source control on wastewater treatment plants, Mississippi River diversion, and flood control in the Upper Mississippi River Basin. We concluded that a suite of practices is needed to effectively deal with hypoxia in the following general categories: (1) modification of farm practices to make the use of nitrogen from fertilizer and manure more effective and efficient; (2) the creation and restoration of wetlands and riparian ecosystems between farmland and streams and rivers, but particularly in those areas where concentrations of nitrate–nitrogen in subsurface drainage is highest; (3) the implementation of nitrogen controls on domestic wastewater treatment plants; and (4) diversion of floodwaters to backwaters of the Mississippi River Delta and coastal wetlands. If policies are devised to implement only one or two of these policies, then improvement in the Gulf of Mexico is not as likely.

We make the following specific recommendations:

1. Several on-farm practices for reducing discharges of nitrogen to streams and rivers should be implemented. These practices, which could lead to 15–20% reductions of nitrogen sources to the Gulf, include a 20% reduction in fertilizer nitrogen application through proper nitrogen crediting for legumes and manure and realistic yield goals. Other recommended management practices include optimum timing of fertilizer application, use of alternative crops, such as perennials, wider spacing of subsurface drains, and better management of livestock manures whether stored or applied to the land.

2. A major effort to restore or create 24 million acres (10 million hectares, or 3.4% of the Mississippi River Basin) of riparian zones and wetlands to reduce nitrogen in the Mississippi River and its tributaries by an average of 40% should be undertaken in the Mississippi River Basin.

3. Wetlands and riparian zones should be strategically placed in watersheds to optimize nitrogen removal as, for example, in tile-drained farmlands that are prone to export high concentrations and amounts of nitrate–nitrogen.

4. Although point sources of nitrate–nitrogen appear to be of little consequence (< 5%) in the overall Mississippi River Basin nitrogen load, an effort to control these sources through tertiary treatment should become a formal policy for new wastewater treatment plants in the basin.

5. The restoration of flood-prone lands in the Upper Mississippi River Basin to wetlands needs to be revisited and more seriously considered in light of the 1993 flood *and* the need to control nitrate–nitrogen to protect the Gulf.

6. Nitrate reduction should be an important consideration in the design and operation of diversions of the Mississippi River for flood events in the Mississippi Delta in Louisiana. Approximately 400,000 to 1 million hectares (1–2.5 million acres) or more of inshore coastal areas (forested wetlands, marshes, and water bodies) should be used for nitrate reduction in diverted waters. An important additional benefit of such diversions would be to address the land-loss problem in Louisiana.

7. Further reductions beyond those now being implemented through the authority of the Clean Air Act are probably not warranted for controlling stationary and mobile atmospheric emissions of nitrogen, at least insofar as protecting the Gulf of Mexico is concerned.

8. There is a strong need for any nitrogen mitigation effort to be coupled to a comprehensive program of monitoring, research, and modeling to evaluate which practices are effective and why, and to allow for "adaptive management" of the hypoxia problem.

CHAPTER 1

Introduction

1.1 THE GULF OF MEXICO HYPOXIA

The main focus of this report is the identification and evaluation of methods to reduce nutrient loads from the continental United States—particularly from the Mississippi, Missouri, and Ohio River basins—to surface and ground waters and, ultimately, the Gulf of Mexico. For at least the past 10 years, seasonally severe and persistent hypoxia (low dissolved oxygen conditions) has been measured on the continental shelf of the northern Gulf of Mexico to the west of the Mississippi and Atchafalaya River deltas. The hypoxia zone has ranged from 13,000 to 20,000 km^2 from 1993 through 1999 (Rabalais et al. 1996, 1997, 1999; Rabalais personal communication). The hypoxia appears to be most widespread, persistent, and severe in June, July, and August (Rabalais et al. 1996). There also appears to be spatial and temporal variability in the distribution of the hypoxia on the shelf, which is, in part, related to the amplitude and timing of the Mississippi and Atchafalaya stream flows.

The waters that discharge to the Gulf originate in the combined Mississippi/Ohio/Missouri watersheds (referred to as the Mississippi River Basin in this report). In total, these watersheds encompass about 3,000,000 km^2 (1,200,000 mi^2), or about 40% of the area of the lower 48 states (Figure 1.1). Two-thirds of the flow from this system enters the Gulf through the Mississippi River, while the remaining one-third enters through the Atchafalaya River. The Mississippi River Basin accounts for 90% of the freshwater inflow to the Gulf (Rabalais et al. 1996).

Linkages between the freshwater inflow from the Mississippi/Atchafalaya River systems (and subsequent nutrient flux) and net surface productivity and bottom-water oxygen deficiency have been generally established (Atwood et al. 1994; Justić et al. 1995; Rabalais et al. 1996) and are discussed in detail in other reports in this series (Rabalais et al. 1999; Diaz and Solow 1999; Brezonik et al. 1999). Freshwater discharge and nutrient fluxes from the Mississippi and Atchafalaya Rivers appear to influence the distribution and intensity of the hypoxia, along with water column stratification and mixing (Rabalais et al. 1991).

The hypoxia zone in the Gulf of Mexico is characterized by increased primary production in the upper water column. Oxygen-demanding organic carbon derived from this primary production sinks, decomposes, and leads to the seasonally severe oxygen depletion in the lower waters and sediments (Turner and Allen 1982; Rabalais et al. 1991, 1992; Bierman et al. 1994; Justić et al. 1996, 1997). The low oxygen causes the benthic community to be characterized by limited species; reduced abundance, species richness, and biomass; and domination by pollution-tolerant organisms. Effects of hypoxia on fishery resources, covered in a companion report (Diaz and Solow 1999) could include direct mortality of fish and their food base, as well as indirect effects, such as altered migration, reduction in suitable habitats, increased susceptibility to predation, and
disruption of spawning, recruitment, and migration.

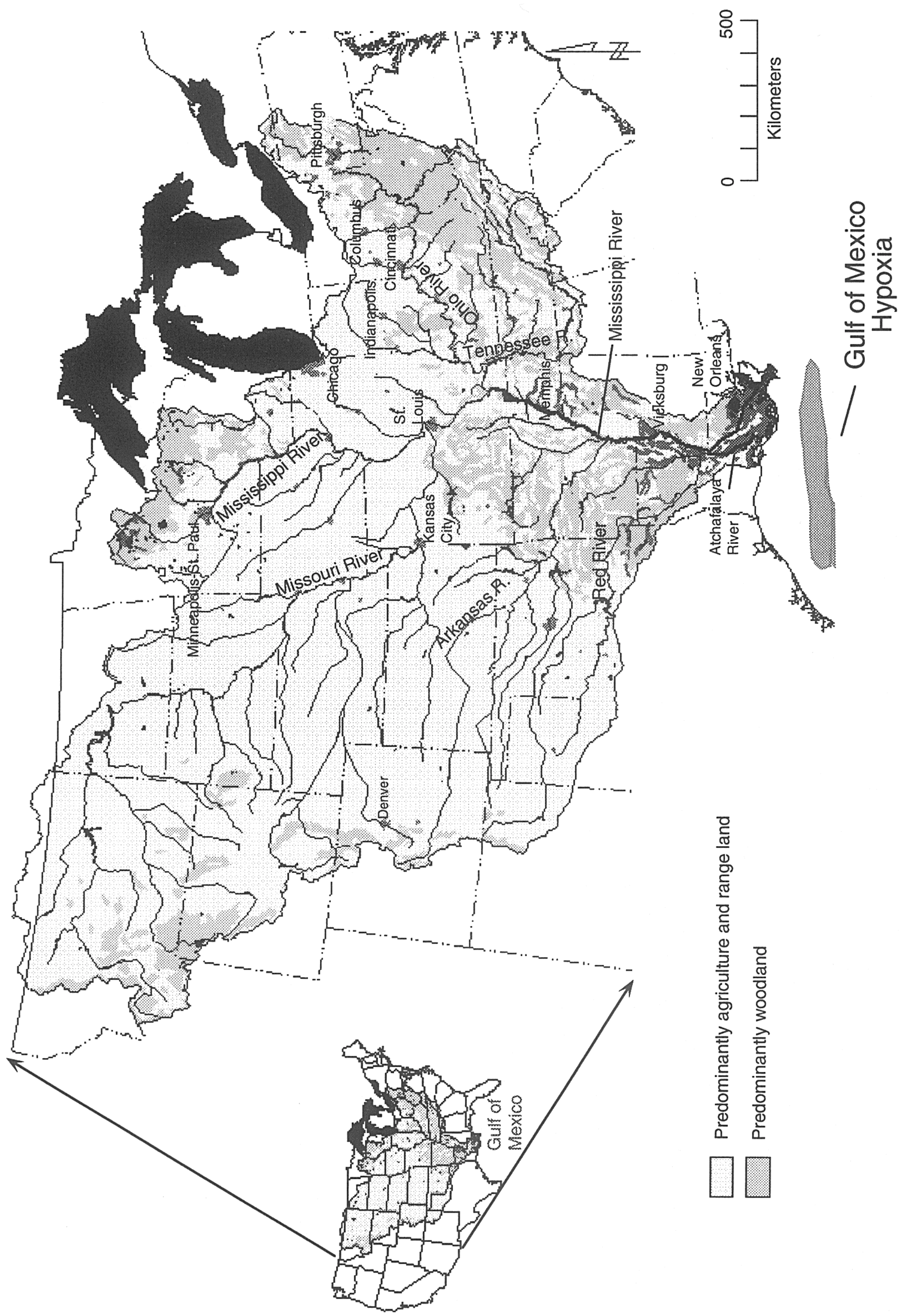


FIGURE 1.1. The Mississippi River Basin and location of the Gulf of Mexico hypoxia. (*Adapted from Goolsby et al. 1999*.)

1.2 OBJECTIVES

It is the goal of this report to identify and evaluate approaches for solving the problem of the hypoxia in the Gulf of Mexico. To accomplish this goal, we have the following objectives:

- review appropriate literature on methods for controlling nutrients, particularly nitrate–nitrogen, from entering waterways;
- evaluate the efficacy of these methods;
- put the methods in the context of the entire Mississippi River Basin and the significance of the sources; and
- give recommendations as to the most reasonable combination of approaches that would be necessary to solve the problem.

1.3 NUTRIENT SOURCES TO THE GULF OF MEXICO

Nutrient loadings, particularly nitrogen loadings associated with eutrophication of coastal marine systems, are transported via atmospheric, surface flow, and ground-water pathways. Nitrate–nitrogen concentrations in the Mississippi River have increased dramatically in this century, and have accelerated since 1950, coinciding with increasing fertilizer use in the Mississippi Basin (Turner and Rabalais 1991; Figure 1.2). Other factors—such as artificial drainage and other changes to the hydrology of the Midwest, atmospheric deposition of nitrates within the Mississippi River Basin, nonpoint discharges from urban and suburban areas, and point discharges, particularly from domestic wastewater treatment systems and feedlots—all contribute to the nutrients that reach the Gulf of Mexico. Table 1.1 presents estimates of the relative inputs of these sources. Controlling these sources through agricultural management, environmental technology, and ecotechnology is the focus of this report. The sources are briefly discussed in the following subsections.

TABLE 1.1. Sources of nitrogen (N) and phosphorus (P) in the Mississippi River Basin (MRB) and export from the basin via the river.

Sources and Output	Total N (Thousands of metric tons/yr)	Total P
Sources to MRB		
Fertilizer Use	6,578	1,020
Mineralized Soil Nitrogen	6,463	0
Legume N-fixation	4,150	0
Feedlots/Manure	2,665	?
Atmospheric Deposition	1,221	0
Point Sources—Municipal	200	30
Point Sources—Industrial	70	28
Urban Nonpoint Sources	?	?
Output to Gulf of Mexico		
From Mississippi River	1,568	136
From Atmosphere	15	0

[1]Sources should not be added, as that would lead to double accounting of some nutrients.
Source: Goolsby et al. 1999.

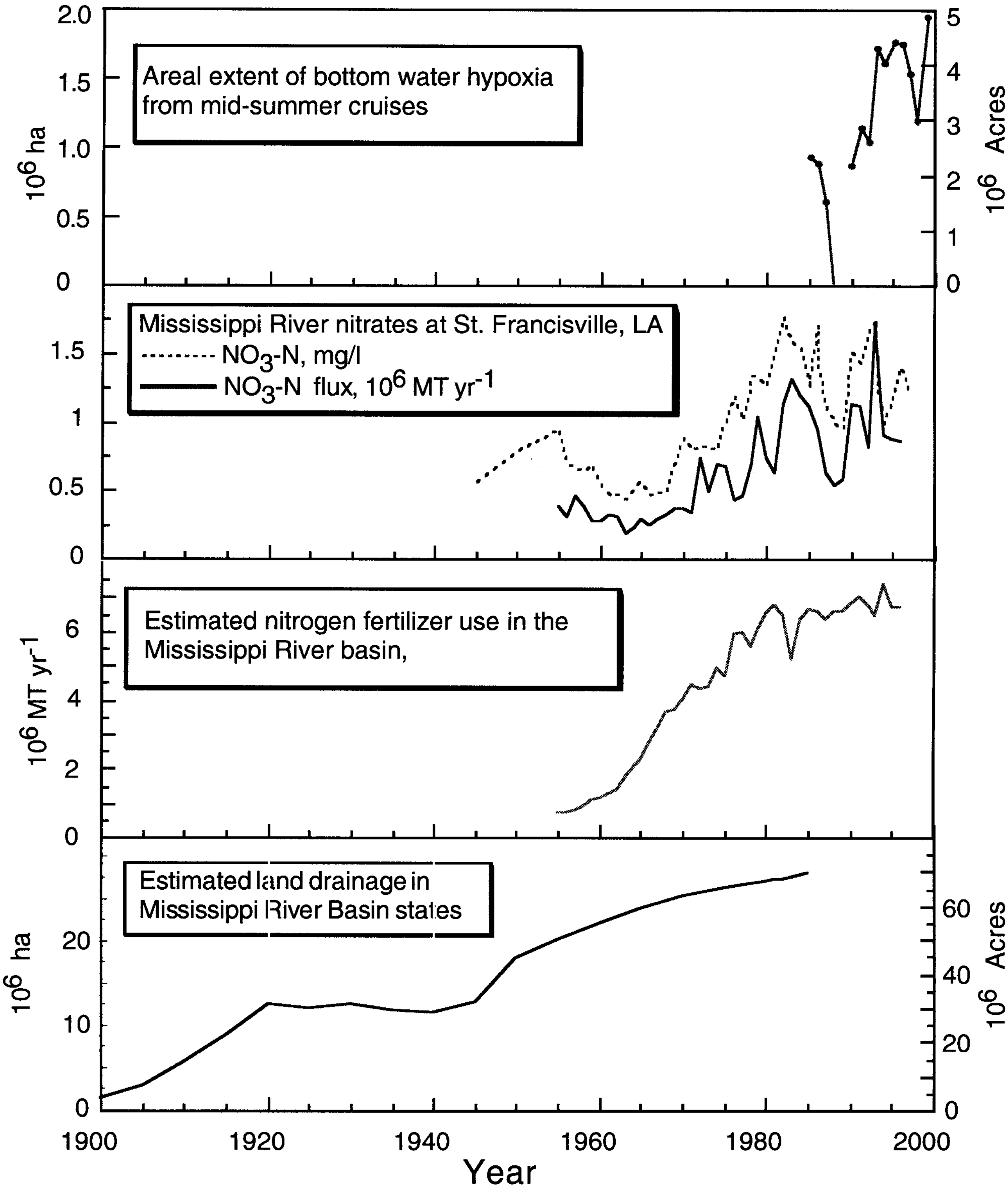


FIGURE 1.2. Estimated trends in the 20th century of the hypoxia area, nitrogen concentrations and fluxes, nitrogen fertilizer use, and land drainage in the Mississippi River Basin. (*Data from Rabalais et al. 1999, Goolsby et al. 1999, and USDA 1987.*)

1.3.1 Crop Production and Soil Drainage

Nitrogen (N) is a naturally occurring element that is essential to plant growth and crop production. Agriculture has been identified frequently as a major contributor of nitrate–nitrogen to surface water throughout the developed world. Omernik (1977) reported that total nitrogen concentrations were nearly nine times greater downstream from agricultural lands than downstream from forested areas, with the highest concentrations being found in the Corn Belt states of the Upper Mississippi Basin. As stated by Power et al. (1998), "the global nitrate problem is most apparent in the North Central region of the United States where 83 percent of the nation's corn is produced and 53 percent of the commercial nitrogen fertilizer is used." Nitrate–nitrogen is continually supplied to streams and rivers through mineralization of soil organic matter, particularly where tile drainage has exposed formerly wet soils to oxidation and through the application of fertilizer and animal manures to cropland. Goolsby et al. (1999) estimate these two sources (fertilizer and mineralization) contribute about 13 million metric tons per year of nitrogen to the Mississippi River Basin (MRB) (Table 1.1). Agro-industrial wastes, atmospheric deposition of volatilized ammonia from manure and fertilizer, and dinitrogen fixation, all of which either occur as nitrate–nitrogen or can be converted to nitrate–nitrogen through mineralization and nitrification, are other important sources. Nitrogen fixation by legumes contributes 4 million metric tons per year of nitrogen to the basin (Table 1.1).

Nitrate–nitrogen is mobile and, therefore, can be lost from the soil profile by leaching. Subsequent transport of nitrate–nitrogen to surface waters occurs primarily through subsurface drainage (tile lines) or base flow. Subsurface drainage is a common water management practice in highly productive agricultural areas of the MRB, where poorly drained soils have seasonally perched water tables or shallow ground water. Very little nitrate–nitrogen is lost from the agricultural landscape via surface runoff (Jackson et al. 1973; Logan et al. 1994).

Several long-term studies on rivers of different stream order draining widely different scales of watershed basins all point to the fact that agricultural practices do affect the nitrate–nitrogen concentrations in river water. Nitrate–nitrogen concentrations in stream water collected from water years 1984–93 for a portion of the Upper MRB were significantly greater (2–6 mg-N/L) from rivers that drain a large percentage of agricultural land compared to rivers that drain a larger percentage of forested land (0.1–0.5 mg-N/L) (Kroening 1996). For perspective, the national drinking water standard of nitrates is 10 mg-N/L. In the Mississippi River, mean concentrations were significantly greater (1.8–2.5 mg-N/L) downstream of the confluence with the Minnesota River (an agricultural watershed) than upstream (0.2–0.9 mg-N/L). Keeney and DeLuca (1993) examined nitrate concentrations in the Des Moines River in 1945, 1955, 1976, and 1980–90 and found the average nitrate–nitrogen concentrations to have changed little in the last 45 years (5.0 mg-N/L in 1945 and 5.6 mg-N/L in 1980–90). They concluded that intensive agricultural practices that enhance mineralization of soil nitrogen, coupled with subsurface artificial drainage, are the major contributors of nitrate–nitrogen to streams and rivers of the Midwest.

Somewhat similar conclusions were drawn by David et al. (1997), who surmised that agricultural disturbance leading to high mineralization rates and nitrogen fertilization combined with subsurface tile drainage contributed significantly to nitrate export in the Embarras River in Illinois. In their six-year study, an average of 49% (with a range of 25–85%) of the large pool of nitrate–nitrogen remaining after harvest was leached through drainage tiles and exported by the river. Precipitation exerted a large influence on drainage losses, with a few days of high-flow events leading to most of the annual loss in some years. Rivers with higher concentrations of nitrate–nitrogen seem to be surrounded by landscapes with similar general characteristics: (1) humid/high rainfall conditions;(2) soils high in organic matter; (3) poorly drained, fine-textured soils needing artificial subsurface drainage for optimum crop production; and (4) domination by corn- and soybean-intensive agriculture.

Soils high in organic matter can mineralize a substantial amount of nitrate-N that is susceptible to loss in subsurface tile drainage, especially when wet years follow very dry years. Tile drainage from continuous corn plots that received only 20 kg N ha^{-1} yr^{-1} at Lamberton, Minnesota, contained annual flow-weighted nitrate-nitrogen concentrations of 13 to 19 mg-N/L (Gast et al., 1978). After an extremely dry year followed by a year with slightly above-normal rainfall, nitrate-nitrogen concentrations averaged 28 mg-N/L from these plots.

In a study at Waseca, Minnesota, four plots were fallowed (no crop grown and no fertilizer applied), with periodic tillage each year from 1987 through 1993. Nitrate–nitrogen concentrations in the tile drainage water averaged 57 mg-N/L in 1990 following three dry years. Concentrations dropped to 38, 25, and 23 mg-N/L in 1991, 1992, and 1993, respectively (Randall, unpublished data). Hatfield (1996) found that nitrate–nitrogen concentrations in the Walnut Creek watershed in Iowa ranged from 15 to 20 mg-N/L throughout most of the year and stated that this loss was due primarily to the high organic matter content of the soils and their ability to mineralize nitrogen. Elevated levels of nitrate–nitrogen will be lost to drainage water in these tile-drained soils, regardless of fertilizer management practices, especially in wet years following dry years when crop production was limited.

1.3.2 Feedlot Discharges

Dairy, cattle, swine, poultry, and aquaculture systems can cause significant discharges of oxygen-demanding substances and nutrients to local streams and rivers. Untreated wastewater from these systems generally has very high concentrations of phosphorus and nitrogen (Table 1.2), the latter most often as ammonia–nitrogen, although high concentrations of nitrate–nitrogen are also possible. Estimates range widely as to the importance of this source in the total nutrient loading to the MRB. Goolsby et al. (1999) found 2.7 million metric tons per year of nitrogen being discharged into the basin, or about 40% of the total fertilizer use in the basin (Table 1.1). Care needs to be taken not to double count nutrients, as some amount of the fertilizer in crop production ends up as effluent in feedlots.

TABLE 1.2. Contrast of nutrient concentrations in secondary effluent from a municipal wastewater treatment plant, wastewater from a confined animal feeding operation, runoff from croplands of the midwestern U.S., and urban (residential) runoff.

Nutrients	**Secondarily Treated Effluent**	**Confined Animal Feeding Operations**	**Corn Belt Cropland**	**Urban (Residential) Runoff**
Suspended Solids (mg/L)	5–20	585	50–1,000	228
Total P (mg/L)	6.8	24	0.14	0.5
Total N (mg/L)	15.8	254	4.4	2.0
Soluble inorganic N (mg-N/L)	8.4	–	3.4	1.8
Ammonia (mg-N/L)	–	122	–	–
N:P ratio	**2.4**	**10.6**	**31.4**	**4**

Sources: Baker 1992; USEPA 1993; and CH2M-Hill 1997.

1.3.3 Other Nonpoint Sources

Urban and suburban areas have significant runoff from lawns, parking lots, rooftops, roads, highways, and other impervious and semi-impervious sources. Goolsby et al. (1999) were unable to provide accurate estimates of nitrogen and phosphorus in urban nonpoint runoff in the MRB, as such accounting or monitoring systems do not exist. Concentrations and fluxes of nutrients, particularly nitrogen and phosphorus, are generally low compared to nonpoint agricultural sources. Concentrations of total nitrogen are generally half or less in urban runoff compared to agricultural land runoff (Table 1.2), and fertilized agricultural land covers a much greater area in the MRB than do urban and suburban land (Figure 1.1).

1.3.4 Point-Source Discharges

Point-source discharges of nitrogen (N) are estimated to add 0.27 million metric tons per year of nitrogen to the streams and rivers of the MRB, or about 1.5% of the total loading generally coming from agricultural lands (fertilizer use, mineralizing soil, and legume N-fixation) (Goolsby et al. 1999; Table 1.1). The major point sources of direct discharges of nutrients, particularly nitrate–nitrogen, appear to be domestic wastewater treatment plants. Conventional wastewater treatment, through secondary treatment, involves removal of suspended materials, pathogens, and oxygen-demanding organics. Organic matter is converted into inorganic forms, including ammonia–nitrogen, nitrate–nitrogen, and ortho-phosphates. Baker (1992) reported that a conventional wastewater treatment plant effluent has a total N concentration of 16 mg-N/L, a soluble inorganic N concentration of about 8 mg-N/L, and a total phosphorus (P) concentration of about 7 mg-N/L. Domestic wastewater is generally phosphorus-rich, with a much lower N:P ratio than agricultural runoff (Table 1.2).

1.3.5 Atmospheric Sources

The importance of atmospheric sources of nitrogen to the Gulf of Mexico has been difficult to quantify. Nitrogen enters the atmosphere from human and natural sources. In high-temperature combustion, characteristic of the internal combustion engine and fossil-fuel burning electric generating stations, N_2 and O_2 gases are combined to form NO_x (NO and NO_2). NO_2 and airborne nitrates return to the earth's surface with rain, snow, and fog (wet deposition) or as gases and particulate (dry deposition). This nitrogen then enters streams and rivers and/or is retained in terrestrial systems in the same pathways as nitrate–nitrogen fertilizer. Intensive agricultural practices, particularly feedlots where ammonia–nitrogen concentrations are high, also result in ammonia volatilization, which increases local ammonia–N concentrations in the atmosphere. These emissions also return to earth through precipitation and dry fallout. For the Gulf of Mexico, direct deposition of nitrogen from upwind sources—e.g., refineries of Texas, New Orleans, and other urban areas—may contribute some nitrogen to the overall loading of the hypoxic zone.

In 1992, NO_x emissions in the United States were about 23 million metric tons/year (USEPA 1995a). An estimated 1.2 million metric tons/year (or 5% of the total U.S. emission) are deposited in the Mississippi watershed (Goolsby 1999; Table 1.1), or about 18% of the fertilizer input and about 6% of the total nitrogen input to the MRB.

About one-third of the total NO_x emissions in the United States comes from electric utilities. Coal-fired combustion contributed about 90% of estimated electric utilities' NO_x emissions (USEPA 1995a), most in the eastern half of the country. Mobile sources are estimated to contribute more than half of the NO_x emission nationwide. Highway vehicles contribute about one-third of the total NO_x emission, mostly from light-duty vehicles and trucks (including all passenger cars)—the most common types of vehicles. In fact, these vehicles alone comprised almost 22% of national NO_x emissions in 1996 (USEPA 1998a).

NO_x emissions contribute to ozone formation, smog, and acid deposition. In 1996, in a rule promulgated by the U.S. Environmental Protection Agency, NO_x emissions were also recognized for the first time as being a significant source of coastal eutrophication (USEPA 1996a). For example, it was estimated that approximately 27% of the total nitrogen loading to Chesapeake Bay comes from atmospheric sources in the form of NO_x emissions (Linker et al. 1993; Valigura et al. 1994).

CHAPTER 2

Methods

The charge to the committee that authored this report was the following:

> The main focus of this report will be to identify and evaluate methods to reduce nutrient loads to surface water, ground water, and the Gulf of Mexico. The analysis will not be restricted to reduction of sources. It will also include means to reduce loads by allowing the system to better accommodate those sources through, for example, modified hydraulic transport and internal cycling routes.

This report was developed through a series of meetings and subsequent writing assignments by the authors, followed by a compilation, review, and rewriting of the report's sections. Meetings were arranged for the committee in St. Louis, Missouri, on March 26, 1998, and May 21, 1998. The first meeting was held to have a general discussion on the subject, decide on an outline for the report, and assign writing tasks. Most committee members prepared their sections for discussion at the second meeting, which involved initial presentations of the sections by the authors, which was followed by discussion and feedback.

Final versions of manuscripts were submitted by most authors by mid-June. Sections were integrated into this final report through significant editing and review. A preliminary draft was distributed to the committee on September 21, 1998, and a phone conference of the committee was held on October 6, 1998. A second draft of the final three chapters, including the recommendations, was circulated to the committee on October 12, 1998, for final comments. A third draft was prepared based on these comments and was submitted to the National Oceanic and Atmospheric Administration (NOAA) for review on October 15, 1998. The manuscript was reviewed by six extramural reviewers and returned to the committee by mid-February 1999. The reviewers' comments were taken into account in a fourth draft, which was returned to the review team on March 10, 1999.

In this analysis, a full suite of possible methods for reducing nitrogen loading to the Gulf of Mexico was initially considered. Then a shorter list of "more feasible" approaches—both on-farm, between the farm and the streams and rivers, and in and along the Mississippi River basin itself—was compiled based on the following criteria: (1) the significance of the source that was being controlled, (2) the proven effectiveness of the methodology; and (3) the positive ancillary benefits that the methods would have both locally and in the Gulf of Mexico. Methods were chosen that would be effective and generally realistic, within broad social and economic constraints. This shorter list of methodologies was then quantified where possible to give overall quantifiable goals in the series of recommendations. Recommendations were then rechecked to make sure that adequate scientific justification was present in the report.

CHAPTER 3

Results—Approaches for Controlling Nitrogen

Preventing nutrients, particularly nitrate–nitrogen, from reaching the Gulf of Mexico can be accomplished through a number of general approaches and specific techniques (Table 3.1), ranging from modification of agricultural practices to the construction and restoration of riparian zones and wetlands as buffer systems between agricultural lands and waterways. This section provides an overview of each major category of nutrient reduction listed in Table 3.1. This report emphasizes reducing nitrogen, particularly nitrate–nitrogen, in the streams and rivers of the Mississippi River Basin. The focus on nitrogen reduction is based on the strong evidence of cause and effect between nitrate–nitrogen increases and subsequent increases in the hypoxia in the Gulf of Mexico, and on the long-time understanding that coastal waters are generally nitrogen-limited (see Rabalais et al. 1999).

TABLE 3.1. Possible approaches for controlling nitrogen in the Mississippi River Basin.

On-Site Control of Agricultural Drainage Changing Cropping Systems Reducing Nitrogen Fertilizer Application Rates Managing Manure Spreading Managing Time of Nitrogen Application Using Nitrification Inhibitors Change Tillage Methods Increasing Drainage Tile Spacing **Off-Site Control of Agricultural Drainage** Wetlands Riparian Zones Controlled Drainage	**Urban Nonpoint-Source Control** Stormwater Runoff On-Site Sewage Disposal **Point-Source Control —Municipal Wastewater and Feedlot Wastewater** Environmental Technology Ecotechnology **Control of Atmospheric NO_x** Stationary-Source Control Mobile-Source Control **Mississippi Delta Diversions** **Upper Mississippi River Flood Control and Restoration**

3.1 ON-SITE CONTROL OF AGRICULTURAL DRAINAGE

The primary factors that influence the nitrate content of surface and subsurface waters draining agricultural landscapes can be divided into two categories—uncontrollable and controllable. Uncontrollable factors include precipitation and other climatic factors. Controllable factors, which include agricultural management practices that can be used by crop producers to best fit the needs of their enterprise, such as: (1) cropping system used, (2) rate of nitrogen applied, (3) time of nitrogen application, (4) placement method, (5) use of a nitrification inhibitor, (6) tillage systems, and (7) tile spacing in subsurface drainage.

Drainage studies can be very useful for assessing the impact of agricultural management practices on surface- and ground-water quality (Hallberg et al. 1986; Kanwar et al. 1987). Subsurface drains integrate the effects of spatial variability and may be a better tool for studying chemical leaching than such methods as porous suction cups and soil cores (Richard and Steenhuis 1988). However, solute concentrations in subsurface drain flow have been shown not to respond immediately to changes in chemical application rates or residual levels in the soil (Jury 1975a, 1975b; Gast et al. 1978; Baker and Johnson 1981). Some time lag is exhibited due to travel time, depending on drain spacing, soil hydraulic properties, and precipitation.

3.1.1 The Role of Precipitation on Agricultural Drainage

Loading of nitrates into surface water is a function of flow and nitrate concentration in the transported water. The amount of subsurface drainage water leaving the landscape is largely a function of climate and soil properties—e.g., precipitation, soil texture, infiltration rate. Drainage is further influenced by the temporal distribution of precipitation within a year and by the amount of annual or growing-season precipitation that occurs. For instance, an 8-cm rainfall in the spring, when evapotranspiration (ET) losses are low and soil moisture in the profile is likely near field capacity, will have a much greater effect on drainage volume than the same rainfall during the middle of the summer, when daily ET losses are high and soil moisture content is far short of field capacity. In the former scenario, storage capacity is minimal and drainage water carrying nitrates is plentiful. A significant soil-water storage reservoir can exist in the soil in the latter scenario, and subsurface drainage may or may not even occur.

3.1.1.1 PRECIPITATION AND DRAINAGE

The effect of climate on subsurface drainage volume is clear in the following subsurface drainage studies. Annual subsurface drainage in an 11-year Minnesota study (Randall and Iragavarapu 1995) with continuous corn ranged 26–618 mm/yr, with an average of 297 mm/yr (Table 3.2). Drainage was least in 1989, when growing-season precipitation was 35% below normal, and greatest in 1991, when growing-season precipitation was 51% above normal. In addition, drainage in a three-year dry period (1987–89) averaged only 43 mm/yr, compared to the following three-year wet period (1990–92), when drainage averaged 549 mm/yr. Similar findings were reported by Weed and Kanwar (1996), who measured tile drainage under both continuous-corn

TABLE 3.2. Influence of precipitation on drainage volume and annual nitrate-N losses.

Year	April–October Precipitation		Nitrate	
	Rainfall[1] (mm)	*Drainage (mm)*	*Conc.[2] (mg-N/L)*	*Lost (kg-N/ha)*
1986	796	402	14	55
1987	586	42	9	4
1988	426	46	15	6
1989	414	26	12	2
1990	789	486	24	112
1991	961	618	24	139
1992	726	417	14	55

[1] 1961–90 normal = 639 mm.
[2] Annual flow-weighted concentration.
Source: Randall and Iragavarapu 1995.

and a corn–soybean rotation on Kenyon–Clyde–Floyd soils in Iowa. Averaged across four tillage systems, drainage in 1991 totaled 244 mm, or 44% above the 1990–92 average (Table 3.3). A six-year study conducted on a Normania clay loam at Lamberton, Minnesota, showed no tile drainage in the first two years, when annual precipitation was 69% and 76% of normal, respectively (Randall et al. 1997). Annual precipitation in those four subsequent years was 95%, 125%, 117%, and 160% of normal, respectively (Table 3.4). These three studies indicate the strong relationship between precipitation and volume of subsurface drainage.

TABLE 3.3. Annual water loss via subsurface tile drainage for cropping systems in Iowa.

Crop System	Annual Water Loss *(cm)*			
	1990	*1991*	*1992*	*Average*
Continuous Corn	18.5	28.0	12.2	19.5
Rotation Corn	14.3	16.7	7.2	12.7
Rotation Soybean	16.0	28.8	11.3	18.7

Source: Weed and Kanwar 1996.

TABLE 3.4. Effect of crop system on amount of subsurface drainage water.

Crop System	Subsurface Drainage Water *(cm)*			
	1990	***1991***	***1992***	***1993***
Continuous Corn	20	178	132	442
Corn–Soybean	18	274	122	488
Soybean–Corn	28	218	175	478
Alfalfa	0	41	56	320
CRP[1]	0	43	86	510
Average of Row Crop Systems	22	223	143	469
Average of Perennial Crop Systems	0	42	71	415
% of Normal Annual Precipitation	95	125	117	160

[1]CRP = Conservation Reserve Program (mixture of grass and alfalfa).
Source: Randall et al. 1997.

3.1.1.2 PRECIPITATION AND NITRATE CONCENTRATIONS

Nitrate concentrations in subsurface drainage water do not appear to vary consistently with daily drain flow but do show seasonal and yearly variability (Kladivko et al. 1991). Factors affecting this variability include crop uptake of N, residual nitrate in the soil from the previous year, and amount and distribution of rainfall. Goolsby et al. (1997) noted that the concentration and flux of nitrate in rivers of the MRB tend to be highest in the spring, when stream flow is highest. These patterns have been noted in several other studies in the Midwest (Keeney and DeLuca 1993; Phipps and Crumpton 1994; Mitsch and Carmichael 1997).

The general effects of precipitation on nitrate concentrations can be illustrated using basin-wide water quality monitoring data collected in the Minnesota River Basin, a 4 million-ha agricultural basin draining to the Upper MRB (Mulla 1997). Mean annual precipitation in the Minnesota River Basin varies from 56 cm on the western side of the basin to 81 cm on the eastern side. The basin is dominated by intensive row-crop agriculture, has soils that generally have organic matter levels greater than 3%, and has subsurface tile drainage on over half of the farmed acreage. Water quality monitoring data from 1977–94 show that nitrate concentrations range from 0.36 mg-N/L in the head waters on the western side to 4.6 mg-N/L at the mouth of the river on the eastern end where it enters the Mississippi River. Mean annual precipitation increases by about 25 cm across this distance, which produces a corresponding and dramatic increase in the discharge from subsurface tile drains into ditches and streams that eventually flow into the Minnesota River. Fewer than 1% of the water quality samples collected since 1977 from the western portion of the basin have a nitrate concentration that exceeds the drinking-water standard of 10 mg-N/L. About 10% of the water quality samples collected over the same period exceed 10 mg-N/L on the eastern side of the basin.

3.1.1.3 PRECIPITATION AND RESIDUAL SOIL NITRATE

Nitrate concentrations and losses are also greatly affected by dry and wet climatic cycles (Randall 1998). Thirty-two tile drainage plots were planted to a corn (16 plots)–soybean (16 plots) rotation from 1987 through 1993 at Waseca, Minnesota. Late each fall after soybean harvest, anhydrous ammonia was applied to four plots at a rate of 150 kg N/ha for corn the following year. Average annual flow-weighted nitrate concentrations and losses from the corn plots are shown in Figure 3.1.

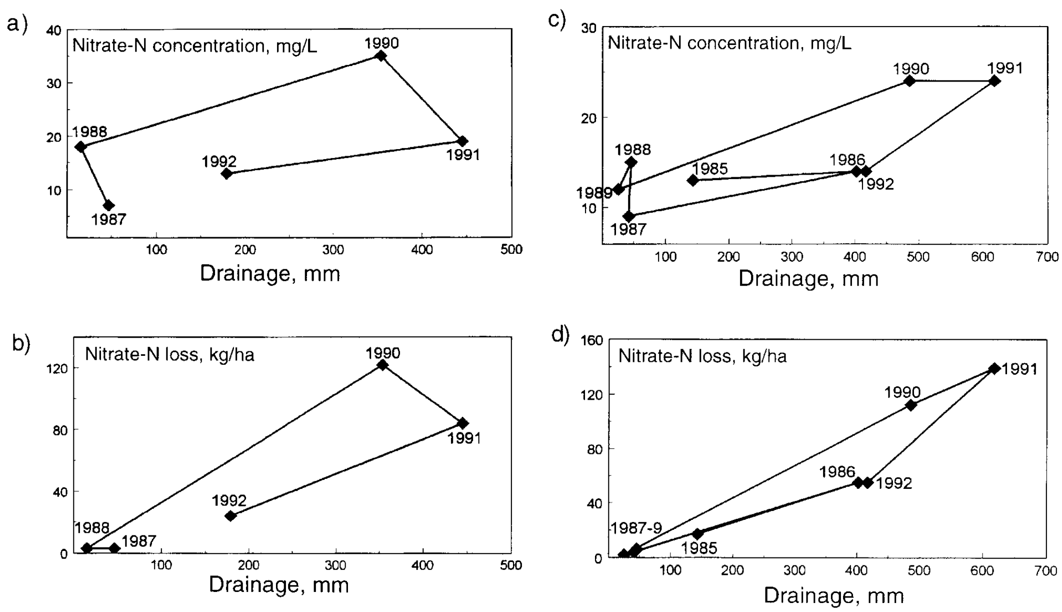


FIGURE 3.1. Relationship between subsurface drainage volume and: (a) annual flow-weighted nitrate–nitrogen concentration; (b) annual nitrate–nitrogen loss in tile-drainage water from a corn–soybean rotation that received 150 kg N/ha as anhydrous ammonia in late October each year following soybeans at Waseca, MN; (c) annual flow-weighted nitrate–nitrogen concentration; and (d) annual nitrate–nitrogen loss in tile-drainage water from continuous corn that received 200 kg N/ha each spring at Waseca, MN.

In 1987 and 1988, when April–October rainfall was 8% and 33% below normal, respectively, subsurface drainage was < 50 mm/yr, and nitrate concentrations ranged between 7 and 18 mg-N/L. Less than 2 mm of drainage occurred in 1989 when April–October rainfall was 35% below normal, and no samples were collected for nitrate–N analyses. Under these dry conditions during the three-year period, corn yields and nitrogen uptake were low. However, residual soil nitrate (RSN) continued to increase in the soil profile to levels as high as 259 kg-N/ha in the top 1.5-m profile following corn. April–October precipitation in 1990 was 23% above normal, causing drainage volume to total > 350 mm. Moreover, annual flow-weighted nitrate concentrations in the corn plots averaged 35 mg-N/L—two times as high as during the dry years (Figure 3.1a). Nitrate–nitrogen concentrations in the soil and drainage water returned to background levels in 1991 and 1992 when rainfall was 50% and 14% above normal, respectively. Nitrate losses from the corn plots showed the combined effect of drainage volume and nitrate–N concentration (Figure 3.1b). These data suggest that RSN can accumulate in the soil profile during dry climatic cycles because of soil mineralization, reduced crop uptake, and every-other-year nitrogen fertilization, even in a corn–soybean rotation. These elevated RSN levels are then poised for delivery to subsurface tile drainage when growing season precipitation returns to above normal.

3.1.1.4 PRECIPITATION AND NITRATE LOADS

In another set of drainage plots at Waseca, nitrogen fertilizer was applied at a rate of 200 kg/ha each spring to corn grown for eight years. Annual flow-weighted nitrate concentrations in 1985 and 1986 averaged 13 and 14 mg-N/L, respectively, although the drainage volume ranged from 143 mm in 1985 to 402 mm in 1986 (Figure 3.1c). Dry conditions during 1987–89, when April–October rainfall was 25% below normal, resulted in < 50 mm drainage/yr and annual average nitrate concentrations ranging from 9 to 15 mg-N/L. RSN totaled 225 kg-N/ha in the 0–1.5 m profile in October 1989. In 1990 and 1991, April–October rainfall averaged 36% above normal and generated annual drainage volumes > 480 mm/yr. In addition, nitrate concentrations in the drainage water doubled from the previous three dry years to 24 mg-N/L in these two wet years. RSN at the end of 1991 was 50% lower than at the end of the dry years. In the third consecutive wet year (1992), more than 400 mm of water drained from the plots, nitrate concentrations in the drainage water returned to 14 mg-N/L, and RSN totaled only 50 kg-N/ha. Nitrate loading in the subsurface drainage water each year was greatly affected by both nitrate concentrations and drainage flow (Figure 3.1d). These data clearly indicate a buildup of RSN in the soil profile during dry years when drainage was limited. Much of the RSN build-up could be attributed to mineralization of soil organic matter, annual additions of N fertilizer, and limited uptake of N by the poor-yielding corn. In the subsequent wet years, substantial losses of nitrate occurred in subsurface drainage due to high concentrations of nitrate and high drainage volumes.

Differences in nitrate contributions across the Minnesota River Basin discussed above in response to gradients in precipitation are even larger when nitrate loads, rather than nitrate concentrations, are compared. Four watersheds located in the wetter eastern portion of the Minnesota River Basin account for 75% of the total nitrate load in the entire basin, yet they drain only 31% of the total basin area. Six watersheds on the drier western side of the basin collectively generate only 7% of the nitrate load. Nitrate yields for watersheds in the Minnesota River Basin average 2.1 kg-N km^{-2} day^{-1} but vary from 0.5 to over 6 kg-N km^{-2} day^{-1}, with the larger yields occurring in the watersheds on the wetter eastern side of the basin.

In summary, precipitation has a great effect on the export of nitrate–nitrogen from crop fields in the MRB, particularly on the nitrate–nitrogen discharged through subsurface drains. Dry years can result in very low discharges of nitrate–nitrogen. Wet years, particularly if they follow one or two dry years, can result in very high discharges of nitrate–nitrogen through subsurface drains.

However, nitrate–nitrogen concentrations do not vary consistently with daily flow, but rather do vary seasonally. Highest concentrations are generally in the spring after spring rains. Between rain events, residual soil nitrate can accumulate in the soil due to fertilization, reduced crop uptake, and soil mineralization, only to be released during high rainfall events.

3.1.1.5 LONG-TERM CHANGES IN PRECIPITATION

Long-term changes in annual precipitation due to climate shifts could have a major effect on nitrate loading to the Gulf of Mexico from the MRB. Increased amounts of annual precipitation would most likely lead to greater surface runoff and subsurface drainage of water containing nitrates. Since 1910, precipitation has increased by about 10% across the contiguous United States, largely due to heavy and extreme precipitation events (Karl and Knight 1998). For extremely heavy precipitation events (> 50 cm), an increasing intensity is also significant. Kunkel et al. (1999) studied the trends of extreme precipitation events and found lengthy periods of below-average numbers of events in the 1930s and 1950s and an above-average number of events in the early 1940s, early 1980s, and 1990s. The overall trend covering the period 1931–96 has been upward at a highly statistically significant rate in a broad region from the central Great Plains across the middle Mississippi River and the southern Great Lakes. The national trend is upward at a rate of 3% per decade for this period.

3.1.2 Changing Cropping Systems

Nitrate concentrations in subsurface drainage water are related to cropping systems. Tile-drainage water from row-crop systems (continuous corn and a corn–soybean rotation) that were fertilized with nitrogen based on a soil nitrate test averaged nitrate–nitrogen concentrations between 14 and 40 mg-N/L from 1990 to 1993 at Lamberton, Minnesota (Table 3.5). In comparison, perennial crops (alfalfa and a conservation reserve program (CRP) grass–alfalfa mix) resulted in nitrate concentrations ranging from 0.3 to 4 mg-N/L. Due to higher-flow volumes from the plots planted to row crops, nitrate losses from the row crops were 30–50 times higher than from the perennial crops (Randall et al. 1997).

TABLE 3.5. Effect of crop system on flow-weighted annual nitrate–N concentrations and four-year total nitrate-N loss.

Crop System	**Annual Nitrate–Nitrogen Concentration** (*mg-N/L*)				**Four-Year Total Nitrate Loss** (*kg-N/ha*)
	1990	*1991*	*1992*	*1993*	
Continuous Corn	30	39	40	20	217
Corn-Soybean	22	29	26	14	204
Soybean–Corn	26	38	27	13	202
Alfalfa	–	4	4	1	7
CRP [1]	–	4	1	0.3	4

[1]CRP = Conservation Reserve Program (mixture of grass and alfalfa).
Source: Randall et al. 1997.

Nitrate concentrations under alfalfa were also shown to be much lower compared to corn or soybeans in Iowa (Baker and Melvin 1994). These findings are similar to those reported by Logan et al. (1980), who found highest nitrate losses with corn, intermediate with soybean or systems where other crops were in rotation, and lowest with alfalfa. Weed and Kanwar (1996) found higher nitrate losses from plots planted to continuous corn compared to a corn–soybean rotation in Iowa (Table 3.6). A four-year field study on a poorly drained, fine-textured soil in northwestern Ohio showed concentrations of nitrate with soybeans were as high as or higher than with corn in a corn–soybean rotation, especially in the spring (Logan et al.

1994). That study concluded that a significant portion of the nitrate in tile drainage is due to nitrogen carried over from the previous corn crop.

TABLE 3.6. Average nitrate concentration and annual nitrate loss in subsurface, tile drainage water in Iowa as a function of crop and tillage technique

Crop Rotation and Tillage	Nitrate Concentration (*mg-N/L*) 1990	1991	1992	Average	Nitrate-Nitrogen Loss (*kg-N/ha*) 1990	1991	1992	Average
Continuous Corn								
Moldboard plow	64	34	12	37	58	63	13	45
Chisel plow	55	28	10	31	100	76	13	63
Ridge tillage	44	21	–	–	83	68	–	–
No tillage	39	19	8	22	107	62	12	60
Corn–Soybean								
Moldboard plow	39	24	8	24	41	36	6	28
Chisel plow	33	21	7	21	51	36	5	31
Ridge tillage	24	19	3	15	34	30	3	22
No tillage	19	17	8	15	32	31	4	22

Source: Weed and Kanwar 1996.

In summary, these studies show substantially higher nitrate concentrations in subsurface drainage from row crops, especially continuous corn, compared to perennial crops that have an extended period of greater root activity (water and nutrient uptake) and where cycling of nitrogen is optimized. Thus, some control of nitrogen losses is possible by changing cropping systems.

3.1.3 CONTROLLING NITROGEN FERTILIZER APPLICATION RATES

Applying the proper rate of nitrogen for a crop is a major management decision facing crop producers. Using too little nitrogen for a highly responsive crop such as corn or wheat results in lower yields, poorer grain quality, and reduced profits. When too much nitrogen is applied, crop yields and quality are not affected, but profit can be reduced somewhat and negative environmental consequences most likely will result. Thus, many farmers choose to err on the liberal side when making decisions on nitrogen rates. This "extra" nitrogen is often called "insurance" nitrogen. The application rate of this excess nitrogen, while difficult to find precisely in the peer-reviewed literature, is stated by one publication to vary between 22 and 67 kg-N/ha (20–60 lb/acre) of excess nitrogen in Minnesota (Legg et al. 1989).

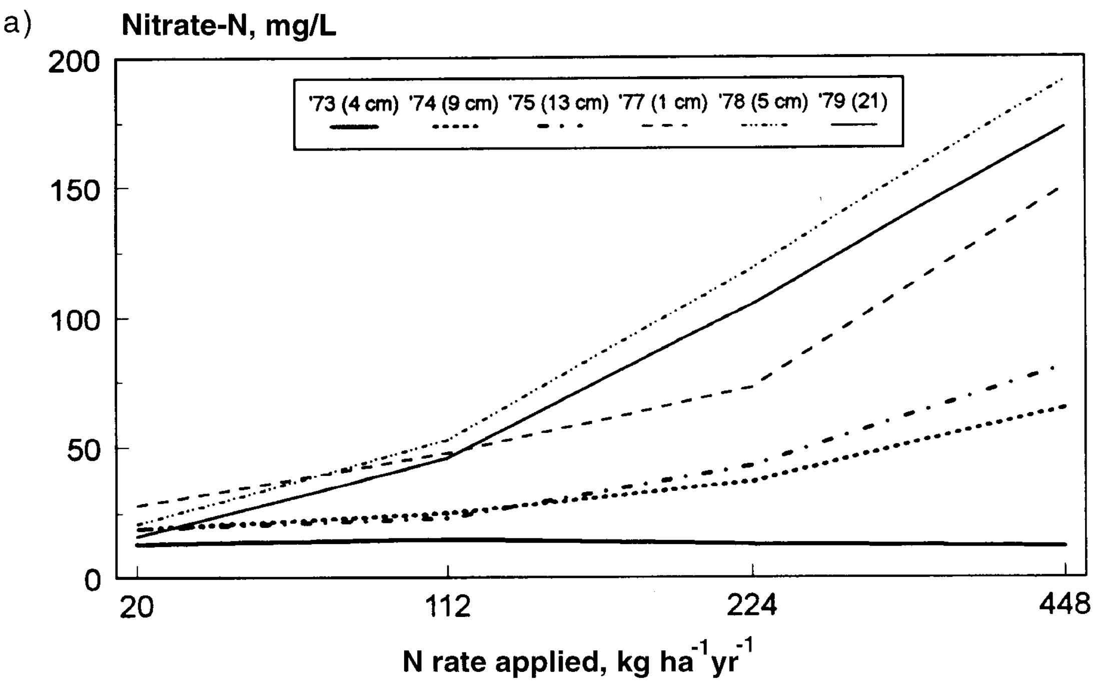
University long-term research provides guidance necessary to make decisions about nitrogen application rates. The recommended application rate provided via various extension bulletins and software venues are based on numerous field experiments conducted across a broad range of soils, cropping systems, and weather conditions. The recommendations also include credits for nitrogen from other sources, such as manure and nitrogen fixed by legumes. These nitrogen credits are then subtracted from the total amount of nitrogen required by the crop to provide a fertilizer nitrogen rate recommendation. Even though the examples used in the following discussion focus on nitrogen fertilizer, it should be remembered that these principles also relate to nitrogen supplied by manure and legume fixation.

The relationship between the annual fertilizer nitrogen application rate for continuous corn and annual flow-weighted nitrate concentrations in subsurface drainage water is shown for two studies in southern Minnesota in Figure 3.2. Climatic conditions during the six-year period (1974–79) at Lamberton, Minnesota, were marked by drier-than-normal conditions, especially during each growing season. Although corn yields were below normal in five of six years with no yield in 1976—a drought year—they were optimized at the 112 kg-N/ha rate in four of five years. Consequently, nitrate concentrations in subsurface drainage water were extremely high with the 224 and 448 kg-N/ha rates, especially in the last three years of the study (Figure 3.2a).

Nitrate concentrations from the 20 kg-N/ha plots ranged between 16 and 28 mg-N/L, indicating the contributing role of soil mineralization in this highly organic soil. Average nitrate concentrations in the drainage water for the two-year pre-drought and the three-year post-drought periods were increased by 16 and 50 mg-N/L when the nitrogen application rate was increased from 112 to 224 kg-N/ha (100–200 lb/acre), respectively. Thus, if a grower decided to apply an extra 45 kg-N/ha (40 lb/acre) of "insurance N" to the recommended 135 kg-N/ha (120 lb/acre) rate for a total of 180 kg-N/ha (160 lb/acre), these data indicate that nitrate concentrations in the tile water would be increased about 6 mg-N/L prior to the drought year and 20 mg-N/L after the drought year. For a six-year period, nitrate concentrations in the drainage water would be expected to increase 14–15 mg-N/L with this extra annual 45-kg "insurance" rate. During this six-year period, if applications of manure yielding about 100 kg-N/ha of available nitrogen annually were not credited, resulting in a total application rate of 235 kg-N ha^{-1} yr^{-1} (135 from the fertilizer and 100 from the manure), the nitrate concentrations would be expected to increase by about 25 mg-N/L. On the other hand, if the annual N fertilizer rate were reduced by about 10% to 125 kg-N/ha and no other nitrogen were applied, one could expect a small yield decrease, and nitrate concentrations could be expected to decrease by about 3 mg-N/L.

At Waseca, the annual nitrogen rates were begun in 1975, but no drainage occurred in 1975 and 1976 due to very dry weather. Thus, at the beginning of 1977 increasingly high amounts of residual soil nitrate remained in the soil profile with each added amount of nitrogen. Consequently, high concentrations of nitrates were found in the 12 cm of drainage water in 1977 (Figure 3.2b).

Nitrate concentrations in the drainage water were lower in 1978 and were further reduced in 1979 as drainage volume increased and yields improved. Annual flow-weighted nitrate concentrations from the control plots (no fertilizer) ranged from 13 to 16 mg-N/L, indicating the role that soil mineralization played during this dry-to-wet climatic cycle in this highly organic soil. Averaged across the three years when tile flow occurred, nitrate concentrations in the drainage water were increased by 16 mg-N/L when the fertilization rate was increased from 112 to 224 kg-N/ha and by 20 mg-N/L when the rate was increased from 224 to 336 kg-N/ha. If 190 kg-N/ha (170 lb/acre) were the recommended nitrogen application rate for a yield goal of 10 metric ton/ha (160 bu/acre), but the grower decided to apply an additional 45 kg-N/ha (40 lb/acre) for "insurance" purposes, based on these data, nitrate concentrations in the drainage water would be projected to increase by about 7 mg-N/L. If an annual nitrogen credit of 100 kg-N/ha from manure were ignored and a total of 290 kg-N/ha were applied annually, nitrate concentrations



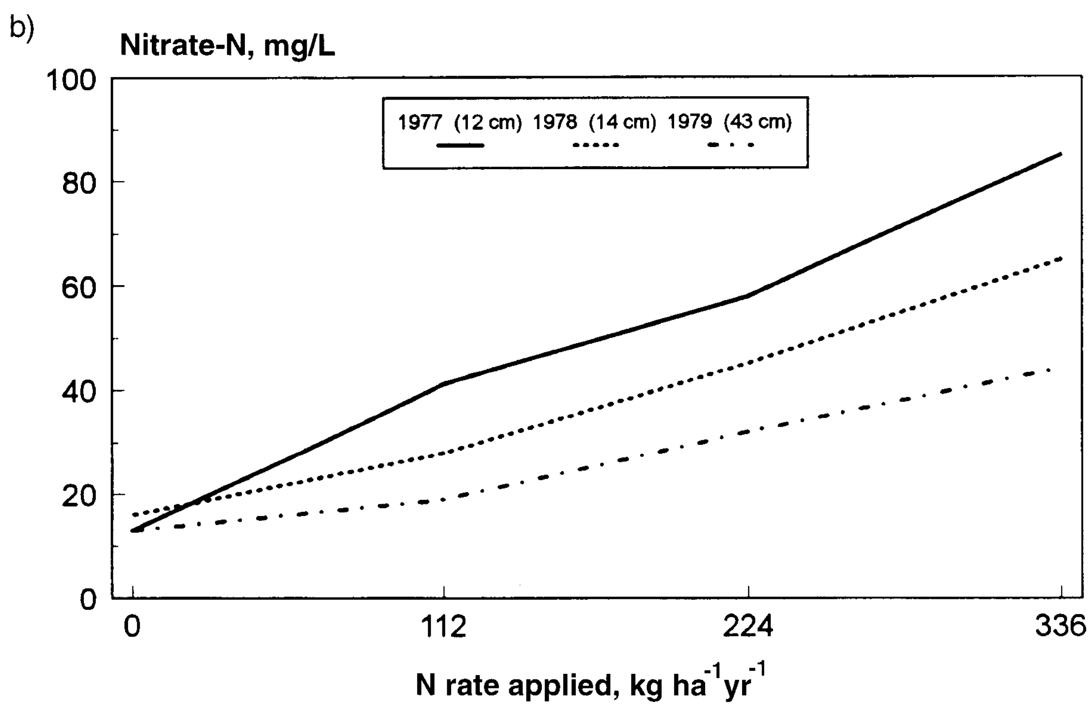


FIGURE 3.2. Nitrate–nitrogen concentration in tile-drainage water as affected by rate of N-fertilizer application for continuous corn at (a) Lamberton and (b) Waseca, MN.

could be expected to increase by about 17 mg-N/L. On the other hand, if the nitrogen fertilization rate were reduced by 10% to 170 kg-N/ha (150 lb/acre), nitrate concentrations could be expected to decrease by about 3 mg-N/L with a relatively low yield reduction (0.3–0.4 metric ton/ha, or 5–6 bu/acre).

Although abnormally dry conditions prevailed for portions of the two above studies, the results clearly show the effect of increasing nitrogen application rates on the concentration of nitrate–nitrogen in tile drainage water. Nitrogen applied in excess of crop need leads to dramatic increases in nitrate concentrations. A simple excess application of 45 kg N/ha for "insurance" purposes can elevate nitrate concentrations by 6–20 mg-N/L, depending on the severity and length of the dry period and on crop yield.

Residual soil nitrate (RSN) that accumulates in the soil profile during dry periods is the major source of the nitrate lost in tile drainage. Accounting for RSN following dry years by using spring soil N tests could be quite helpful to growers (Magdoff et al. 1984; Blackmer et al. 1989; Bundy et al. 1992; Schmitt and Randall 1994). Unless the nitrate has been leached below the top 30 cm, these tests should be able to provide information that would lead to reductions in the recommended rate of nitrogen fertilizer, resulting in lower nitrate–nitrogen losses in subsurface drainage water.

3.1.4 Managing Manure Spreading

Improved manure management, including uniform application of known nutrient amounts and immediate incorporation, is critical if the optimum nitrogen rates are to be achieved in livestock production systems. All too often manure is applied with a disposal objective in mind, rather than with a utilization objective. When this occurs, rates of nitrogen as manure tend to be high and are not distributed evenly across the field. Consequently, credit is not given for nitrogen in the manure, and the total rate of nitrogen application (fertilizer plus manure) becomes excessive. When the nutrient content of manure is known and best management practices are used in land application, manure does not lead to greater nitrate losses to subsurface tile drainage than does nitrogen from commercial fertilizer (Iragavarapu et al. 1997). If manure is applied at greater than agronomic rates, concentrations of nitrate in the drainage water will be elevated.

3.1.5 Managing the Time of Nitrogen Application

The time of nitrogen application is another management decision that crop producers make each year. Agronomically and environmentally speaking, spring is frequently superior to fall application because less nitrogen is lost in the two- to three-month period between application and nitrogen uptake by the crop. However, many corn growers, especially in the northern part of the Corn Belt, prefer applying nitrogen in the fall because they usually have more time in the fall, N-fertilizer prices are often lower, and field conditions are better. In the spring, early planting of corn as soon as the soils are fit is desirable for highest yields and profit. Thus, the window of opportunity for spring N application becomes very narrow (Randall and Schmitt 1998).

N-fertilizer management, particularly managing the rate and time of application, plays a dominant role in the loss of nitrate to surface waters. In one series of experiments, nitrogen was applied in the fall and spring for continuous corn during a sic-year period at Waseca, Minnesota, to investigate the importance of the time of application (Buzicky et al. 1983). In general, nitrate losses from the crop lands were higher when fertilizers were applied in fall. Corn yields were 8% lower and annual losses of nitrates in the subsurface drainage water were 36% higher with a late-fall (early November) application of fertilizer compared to spring application (Table 3.7). Averaged across time of application, yields and nitrate losses in the drainage water were 17 and 30% higher, respectively, for the 202 kg-N/ha fertilizer application rate compared to the 134 kg-N/ha rate. At the end of the study, 65% of the nitrogen being lost in the drainage from high fall treatment was derived from the fertilizer, whereas only 15% of the nitrogen in the drainage water lost from a low spring treatment was derived from the fertilizer (Buzicky et al. 1983).

Based on these data, obtained during a climatic period without a very dry year or years, a 45 kg-N/ha application of "insurance N" above the recommended 190 kg-N/ha rate would increase nitrate losses in tile drainage water by about 6 kg ha^{-1} yr^{-1}. Reducing the optimum nitrogen application rate by 10% and 20% to

170 and 150 kg-N/h, respectively, would most likely reduce nitrate losses by 2.5 and 5.0 kg-N ha^{-1} yr^{-1}, respectively. Corn yields would also most likely be reduced slightly (by 0.3 and 0.7 metric ton/ha (5 and 12 bu/acre) respectively) with the 10% and 20% reductions in the fertilizer application rate.

TABLE 3.7. Effect of nitrogen application rate and time of application on nitrate-N losses and corn yield.

N Application Rates *kg-N/ha*	**Time of Application**	**Annual Loss of Nitrate in Drainage** *kg-N ha^{-1} yr^{-1}*	**Five-Year Average Corn Yield** *metric ton/ha*
0	–	8	4.1
134	Fall	30	8.2
134	Spring	21	9.4
202	Fall	38	10.0
202	Spring	29	10.5

[1]*Ammonium sulfate applied about 1 November or 1 May.*
Source: Buzicky et al. 1983.

3.1.6 Using Nitrification Inhibitors

In another set of experiments (Randall and Vetsch 1995), anhydrous ammonia fertilizer was applied in four treatments with and without a nitrification inhibitor to drainage plots at Waseca, Minnesota (Table 3.8).

TABLE 3.8. Effect of time of N application and nitrapyrin (NI) on nitrate-N losses and corn yield in a corn–soybean rotation during 1990–93.

N Treatment[1]	**Average Annual Flow-Weighted**		**Average Yield**
	NO_3-N conc. mg/L	***Total NO_3-N Lost*** *kg/ha*	*metric ton/ha*
Fall	20	264	8.0
Fall + NI	17	208	8.6
Spring	16	177	8.6
Split	16	190	9.0
Fallow	36	365	–

[1]Anhydrous ammonia applied 25 October (fall) or 1 May (spring).
Source: Randall and Vetsch 1995.

Results obtained from this study and other similar studies suggest that application of anhydrous ammonia in the spring or in late fall along with a nitrification inhibitor (N-Serve) would reduce nitrate concentrations and fluxes in drainage water and increase corn yields, compared to a late fall application of anhydrous ammonia without a nitrification inhibitor. Early fall applications of anhydrous ammonia, when soil temperatures are warmer and conversion to nitrate (nitrification) is faster, would be expected to produce even greater losses of nitrate to drainage water and also poorer yields.

3.1.7 Changing Tillage Methods

Tillage methods appear to have little influence on nitrate losses from agricultural fields. Studies conducted in Iowa showed that tillage methods have less effect on nitrate loss to drainage water than do crop rotations (Bjorneberg et al. 1996; Weed and Kanwar 1996). Moldboard plowing gave the lowest flow volumes, while ridge tillage and no tillage had the lowest nitrate–nitrogen concentrations (Table 3.6).

An 11-year study with continuous corn at Waseca, Minnesota, showed similar results (Randall and Iragavarapu 1995). Although slightly more water drained from the no-till plots, nitrate concentrations were slightly lower compared to moldboard plow plots (Table 3.9). Thus, nitrate flux in subsurface drainage was not significantly reduced by no-till farming practices. Drain flow from corn grown on a loam soil in Ontario was significantly greater for no tillage compared to conventional tillage (CT), while nitrate concentrations tended to be greater with CT (Patini et al. 1996). During a 40-month period, nitrate loss in tile effluent was not significantly different for the two tillage treatments. Thus, nitrate flux in subsurface drainage does not appear to be reduced by no-till practices. This conclusion may not apply to all regions of the Midwest, however, because of changing nutrient management practices. Increased fall tillage has increased nitrate fluxes in subsurface drainage in studies in Iowa (J. Hatfield, personal communication).

TABLE 3.9. Effect of tillage on nitrate losses in subsurface drainage.

Parameter	**Tillage System**[1]	
	Moldboard Plow	***No Till***
Drainage Volume (mm)	279	315
Nitrate-N Conc. (mg/L)	15	13
Nitrate-N Lost (kg/ha)	43	41
N Lost as % of Applied N	21	20

[1]11-year (1982–92) average.
Source: Randall and Iragavarapu 1995.

3.1.8 Increasing Drainage Tile Spacing

Many farmers install additional subsurface drain tile to narrow the spacing between tiles with the expectation that crop yields will be improved because of enhanced drainage. Studies have illustrated that this practice may also increase subsurface losses of nitrates to streams and rivers.

A three-year study on a poorly drained Clermont silt loam soil in Indiana showed drain spacing to markedly affect nitrate losses in the subsurface drainage water (Kladivko et al. 1991). Annual nitrate losses averaged across the three years were 29, 41, and 55 kg-N ha^{-1} yr^{-1} for the 20-, 10-, and 5-m drain spacings, respectively. Crop yield measurements taken during a four-year period at this site showed lower corn yields on the 5-m spacings than on the 10-m or greater spacings (Larney et al. 1989). Averaged across a 10-year period, corn yields were not different among the three drain spacings.

These data suggest the potential for greater residual soil nitrate in the profile due to less crop uptake in dry years, and thus greater leaching losses to subsurface drainage in the following wet year with narrower drain tile spacing. Although few data exist, inferences drawn from this work suggest that narrowing tile drainage to spacings < 20 m could result in greater losses of nitrates compared to wider spacings. Additional research is needed to more clearly define the agronomic and environmental influences of tile spacing in subsurface drainage systems.

3.2 Off-Site Agricultural Nonpoint-Source Control

A second general approach, after on-site approaches, for preventing nitrogen from reaching streams and rivers of the Mississippi River Basin is to place ecosystems that are effective nitrogen sinks between the agricultural fields and the streams and rivers. This section discusses the general functioning of nitrogen in wetlands and riparian systems and then reviews some of the studies and design principles related to using three general ecological systems for controlling nitrogen: (1) natural and created wetlands, (2) riparian buffers, and (3) controlled drainage systems for the control of nonpoint-source pollution, particularly nitrogen, from agricultural fields.

Many of the original freshwater wetlands and riparian zones that were once found throughout the MRB and that were once connected to streams and rivers of the basin are gone from the landscape. Without them, the landscape has lost part of its ability to maintain a biogeochemical balance, and the streams and rivers are no longer buffered from upland regions (Mitsch 1994). The net result has been the loss of a valuable biological habitat and poorer water quality.

For example, in the U.S. Midwest, states such as Ohio, Indiana, Illinois, and Iowa, where over 80% of the wetlands have been drained (partly in response to the Swamp Lands Acts of 1849, 1850, and 1860; Figure 3.3), water quality is particularly degraded as nutrients, pesticides, and sediments from farms and urban areas have nowhere to go except directly into waterways. Seven states in the Upper MRB (Indiana, Illinois, Iowa, Minnesota, Missouri, Ohio, and Wisconsin) collectively have had about 18.6 million ha (46 million acres) of land drained (Table 3.10). Statistics reveal that collectively, these seven states lost the equivalent of 14.1 million ha (35 million acres) over the past 200 years (Dahl 1990). Had natural wetlands and riparian zones been left on the midwestern U.S. landscape, water pollution problems might not be as pervasive and nitrogen fluxes down the Mississippi to the Gulf of Mexico would not have been as extreme.

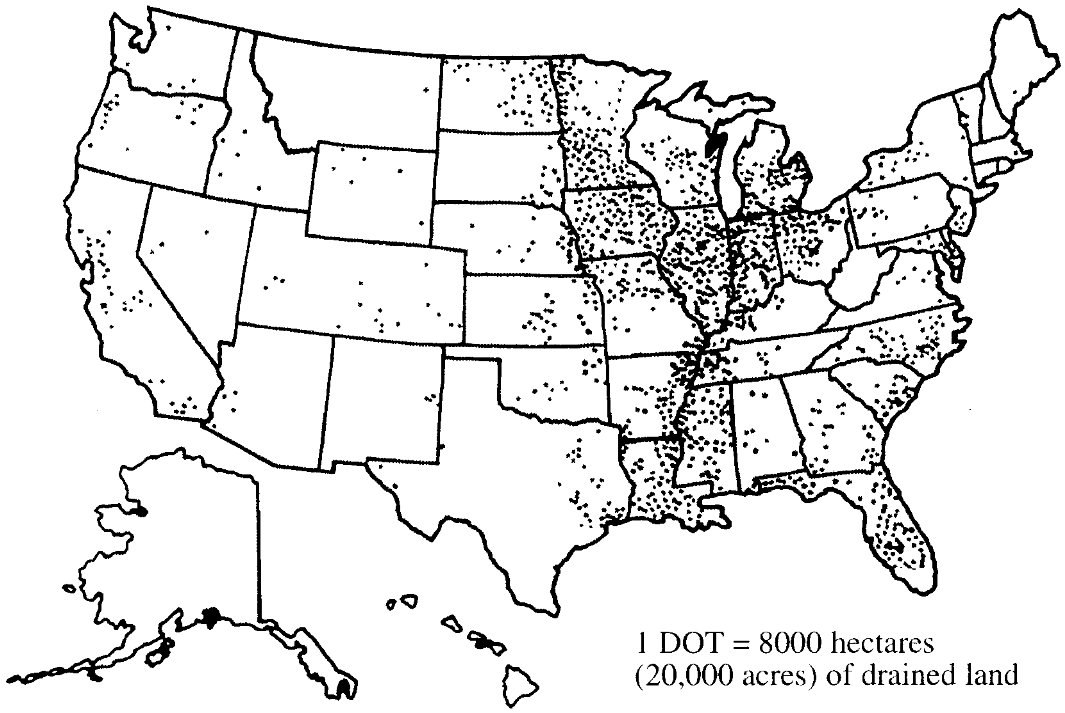


FIGURE 3.3. Extent and location of artificially drained agricultural land in the United States. (*From Dahl 1990*.)

TABLE 3.10. Drainage statistics of selected states in the upper reaches of the MRB.

Basin States	Total Area Drained, x 1,000 ha	% of All Land That Is Drained	% of Cropland That Is Drained
Illinois	3,965	30	35
Indiana	3,273	30	50
Iowa	3,154	20	25
Ohio	3,000	20	50
Minnesota	2,580	15	20
Missouri	1,720	10	25
Wisconsin	910	6	10
Total	**18,602**		

Source: USDA 1987, as cited in Zucker and Brown 1998.

3.2.1 Nitrogen Processes in Wetlands and Riparian Systems

Nitrogen transformations in wetland and riparian soils, surface water, and ground water involve several microbiological processes, some of which make the nutrient less available for plant uptake (Mitsch and Gosselink, 1993; Figure 3.4). The ammonium ion is the primary form of mineralized nitrogen in most flooded wetland soils, although much of the nitrogen can be tied up in organic forms in highly organic soils. The presence of an oxidized zone over the anaerobic or reduced zone is critical for several of the pathways. Nitrogen mineralization refers to "the biological transformation of organically combined nitrogen to ammonium nitrogen during organic matter degradation" (Gambrell and Patrick 1978). This pathway occurs under both anaerobic and aerobic conditions and is often referred to as *ammonification*. Typical formulas for the mineralization of a simple organic nitrogen compound, urea, are given as:

$$NH_2 \bullet CO \bullet NH_2 + H_2O \dashrightarrow 2NH_3 + CO_2$$

$$NH_3 + H_2O \dashrightarrow NH_4^+ + OH^-$$

Once the ammonium ion (NH_4^+) is formed, it can take several possible pathways. It can be absorbed by plants through their root systems or by anaerobic microorganisms and converted back to organic matter. It can also be immobilized through ion exchange onto negatively charged soil particles. Because of the anaerobic conditions in wetland soils, ammonium would normally be restricted from further oxidation and would build up to excessive levels were it not for the thin oxidized layer at the surface of many wetland soils. The gradient between high concentrations of ammonium in the reduced soils and low concentrations in the oxidized layer causes an upward diffusion of ammonium, albeit very slowly, to the oxidized layer. This ammonium nitrogen then is oxidized by a restricted number of chemoautotrophic bacteria through the process of nitrification in two steps:

$$2NH_4^+ + 3O_2 \dashrightarrow 2NO_2^- + 2H_2O + 4H^+ + \text{energy}$$

and,

$$2NO_2^- + O_2 \dashrightarrow 2NO_3^- + \text{energy}$$

Nitrification can also occur in the oxidized rhizosphere of plants, where adequate oxygen is often available to convert the ammonium-nitrogen to nitrate-nitrogen (Reddy and Graetz 1988). Nitrate (NO_3^-), as a negative ion rather than the positive ammonium ion, is not subject to immobilization by the negatively charged soil particles and is thus much more mobile in solution. If it is not assimilated immediately by plants or microbes (*assimilatory nitrate reduction*), or lost through ground-water flow due to its rapid mobility, it has the potential of going through *dissimilatory nitrogenous oxide reduction*, a term that refers to several pathways of nitrate reduction, the most prevalent being reduction to ammonia and *denitrification*. Denitrification, carried out by microorganisms in anaerobic conditions with nitrate acting as a terminal electron acceptor, results in the loss of nitrogen as it is converted to gaseous nitrous oxide (N_2O) and molecular nitrogen (N_2):

$$C_6H_{12}O_6 + 4NO_3^- \dashrightarrow 6CO_2 + H_2O + 2N_2$$

As illustrated in Figure 3.4, the entire process occurs after (1) ammonium–nitrogen diffuses to the aerobic soil layer, (2) nitrification occurs, (3) nitrate–nitrogen diffuses back to the anaerobic layer, and (4) denitrification occurs. Because nitrate diffusion rates in wetland soils is seven times faster than ammonium diffusion rates, ammonium diffusion and subsequent nitrification are thought to limit the entire process of nitrogen loss.

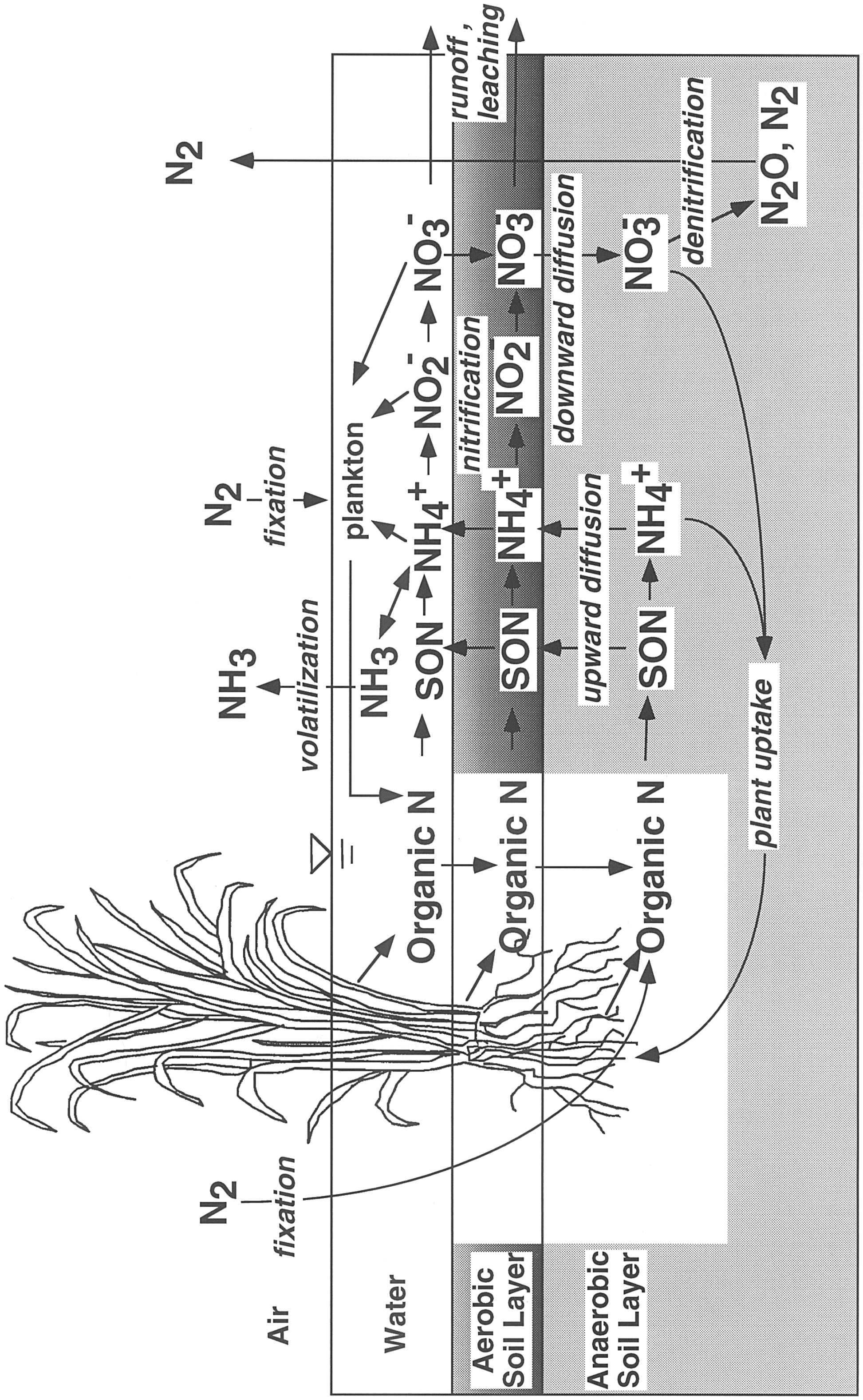


FIGURE 3.4. Nitrogen transformation in wetlands. SON indicates soluble organic nitrogen. (*From Mitsch and Gosselink 1993.*)

3.2.2 Wetlands

Some rates of denitrification and nitrogen retention measured for wetlands or wetland soils in laboratories appear in Table 3.11. If sufficient nitrate and organic carbon are available, high rates of denitrification (> 100 g m^{-2} yr^{-1}) are physically possible. It also appears that on a per unit-area basis, wetlands (swamps, marshes, and possibly peatlands) have a greater potential for nitrate–nitrogen reduction by denitrification than do riparian forests.

TABLE 3.11. Nitrogen loss rates as reported in the literature for wetland and riparian zone studies.

Rates *g-N m^{-2} day^{-1}*	*g-N m^{-2} yr^{-1}*	**Conditions**	**References**
Laboratory Studies			
0.06-0.92		10 mg N-NO_3/L added	Gale et al. 1993
Wetlands			
	0.03–5.5	Natural swamps (LA)	Smith and DeLaune 1983
	28	Nutrient-enriched swamp (FL)	Dieberg and Brezonik 1985
0–0.2		Range for natural wetlands	Nixon and Lee 1986
	6	Discharge fen, The Netherlands	Koerselman et al. 1989
	20	Recharge fen, The Netherlands	
	0.002–0.34 avg. = 0.19	Low-nutrient wetlands	Johnston 1991
	16–134 avg. = 60	N-enriched wetland	
	20–92	Danish wetlands	Jørgensen 1994
	17[1]	River-fed constructed wetlands (IL)	Phipps and Crumpton 1994
	280	Treatment wetland—theoretical rate	Kadlec and Knight 1996
	80[1]	Treatment wetland—based on I/O water quality analyses	
0–3.46		Agricult. runoff—*Phragmites* marshes Marshes in Spain	Comín et al. 1997
	101[1]	Wastewater constructed wetland (OH)	Spieles and Mitsch 2000
	62–66[1]	River-fed constructed wetlands (OH)	
Riparian Systems			
	4.5-–6. 0	Riparian forest, Chesapeake Bay (MD)	Peterjohn and Correll 1984
0.22		NO_3 + glucose; buffer zones	Groffman et al. 1991
1.58		NO_3 + glucose; grass strips	
	0.5–1.6	Riparian maple swamp (unenriched)	Hanson et al. 1994
	2.0–3.6	Riparian maple swamp (enriched)	
	6.9	Restored riparian wetland	Lowrance et al. 1995
	4.3	Young hardwood riparian forest	
0.87–1.3[2]		Moderately well-drained soil	Groffman and Hanson 1997
2.6–24.4[2]		Very poorly drained soil	
	1.5–15.5[2]	Alluvial soil	Groffman and Hanson 1997
	1.0–2.0[2]	Light till	

[1]Net reduction of NO_3 + NO_2 through wetlands. [2]Range of annual means (14–16 samples/yr).
Partial source: Groffman 1994.

Wetlands and riparian zones can be nutrient *sources*, *sinks*, or *transformers*. A wetland is considered a sink if it has a net retention of an element or a specific form of that element (e.g., organic or inorganic)—that is, if the inputs are greater than the outputs. If a wetland exports more of an element or material to a downstream or adjacent ecosystem than would occur without that wetland, it is considered a source. If a wetland transforms a chemical from, say, dissolved to particulate form, but does not change the amount going into or out of the wetland, it is considered to be a transformer. There is not consensus on this question for wetlands in general; in fact, there is little agreement in the literature even for particular nutrients in specific wetland types (Richardson 1985). All that can be said with certainty is that many wetlands act as sinks for particular inorganic nutrients, and many wetlands are sources of organic material to downstream or adjacent ecosystems.

The three types of wetlands (Figure 3.5), whether natural, restored, or created, that could be utilized for the control of nonpoint source pollution in the Mississippi River Basin are: (1) freshwater marshes; (2) peatlands, e.g. bogs/fens; and (3) forested wetlands, including riparian forests.

Table 3.12 lists some of the studies where nitrogen retention has been examined in these types of systems for relatively low concentrations of nitrogen that would be typical of nonpoint-source pollution. There is a dramatic difference in nitrogen concentrations and N:P ratios for treated wastewater and rural nonpoint-source pollution (Table 1.2). Therefore this section will emphasize wetlands that are receiving concentrations typical of nonpoint-source pollution. Discussion of constructed wetlands for wastewater treatment is in Section 3.4.

3.2.2.1 NATURAL FRESHWATER MARSHES

Freshwater marshes are among the most studied types of wetlands for their role in water quality improvement. Early studies by Klopatek (1978) in a Wisconsin riverine marsh and by Simpson et al. (1978) in a tidal freshwater marsh showed the capacity for marsh wetlands to be at least a seasonal sink for inorganic forms of nitrogen and phosphorus. A two-year study of the potential of managed marsh wetland in upper New York State to remove nutrients from agricultural drainage gave inconsistent results, with the wetland acting as a source of nitrogen and phosphorus in the first year and as a net sink in the second year (Peverly 1982). Studies of a freshwater marsh along Lake Erie's shoreline have shown that the wetland is effective in ameliorating nutrient loading from an agricultural watershed to the lake and that the effectiveness is dependent on the amount of annual runoff and the level of the lake (Klarer and Millie 1989; Mitsch and Reeder 1991, 1992).

3.2.2.2 CREATED AND RESTORED MARSHES

Despite the apparent success of natural wetlands to retain some nonpoint-source pollution, it has become more common in the United States to discuss the construction or restoration of new wetlands, rather than the use of natural wetlands when purposeful use of a wetland for nonpoint-source pollution control is discussed (Olson 1992). The construction of new wetlands for controlling nonpoint-source pollution is a more recently proposed approach for wetlands and water quality, although studies have investigated the idea in detail (Livingston 1989; Hey et al. 1989; Mitsch and Cronk 1992; Baker 1992; Mitsch 1990, 1992, 1995), compared to the more abundant literature on natural wetlands and wastewater wetlands. Wetlands built for controlling nonpoint-source pollution (e.g., sediments and nutrients) need to be considered part of a watershed or river floodplain restoration project.

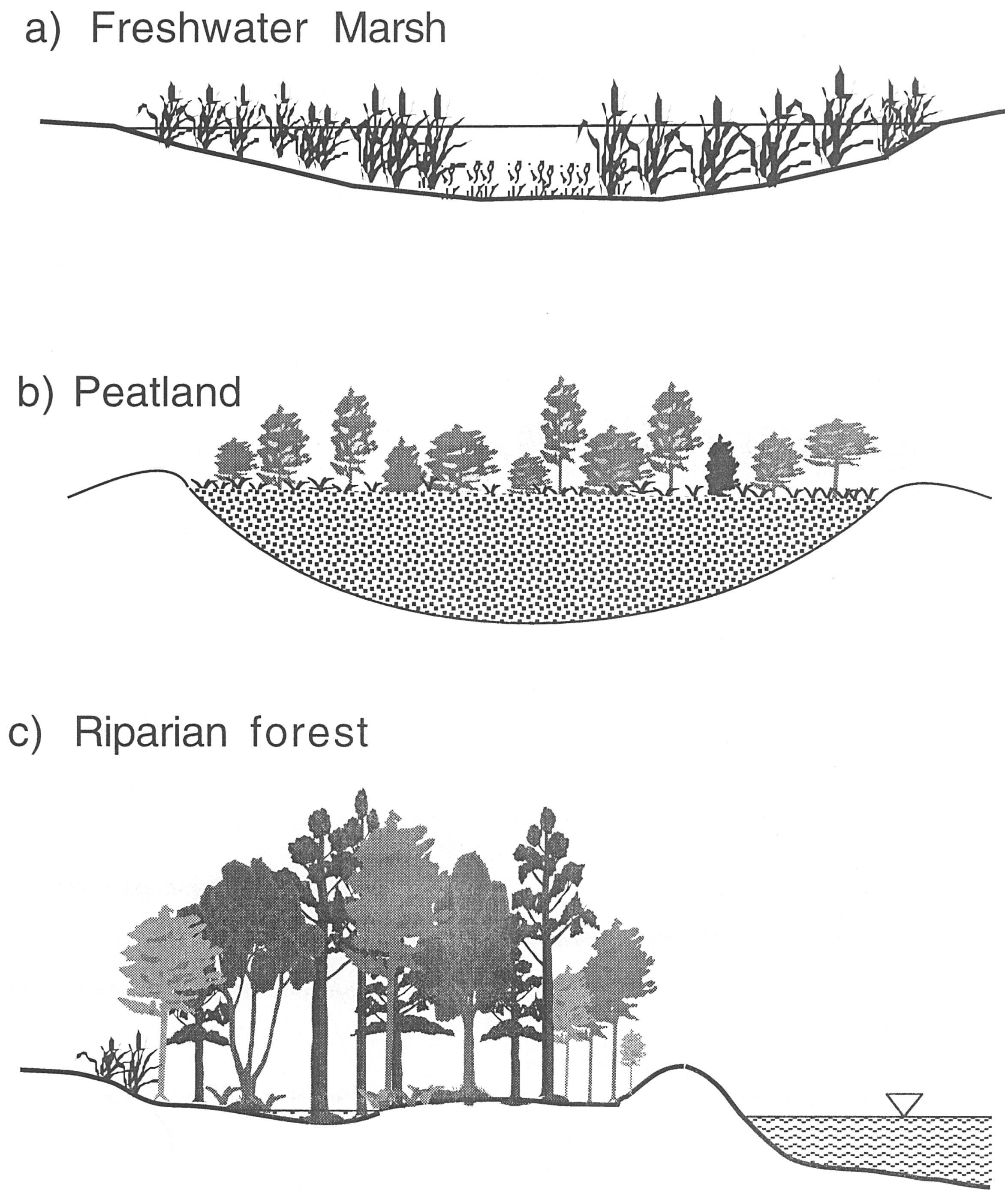


FIGURE 3.5. Three types of wetlands that could be used to control nonpoint-source pollution: (a) freshwater marsh, (b) peatland, and (c) riparian forest.

TABLE 3.12. Selected studies that have investigated nitrogen retention of natural and created freshwater marshes, peatlands, and forested wetlands and riparian zones.

Types and Locations of Wetlands	Nitrogen Sink?	References
Natural Freshwater Marshes		
Four marshes, WI	Yes	Lee et al. 1975
Water hyacinth marsh, FL	Yes	Mitsch 1977
Theresa Marsh, WI	Seasonal	Klopatek 1978
Managed marsh, NY	Inconsistent	Peverly 1982
Natural marshes, Albury, Australia	Seasonal	Raisin and Mitchell 1995
Phragmites marshes, Ebro River, Spain	Yes	Comín et al. 1997
Constructed Freshwater Marshes		
Lake Jackson, FL	Yes	Johengen and LaRock 1993
Des Plaines River wetlands, IL	Yes	Kadlec and Hey 1994 Phipps and Crumpton 1994
Olentangy River wetlands, OH	Yes	Mitsch and Carmichael 1997 Mitsch and Montgomery 1998 Spieles and Mitsch 2000
Boney Marsh, FL	Yes	Moustafa et al. 1996
Constructed marsh, Albury, Australia	Inconsistent	Raisin and Mitchell 1995 Raisin et al. 1997
Peatlands		
Forested peatland, MI	No	Richardson et al. 1978
Thoreau's Bog, MA	Yes	Hemond 1980
Black spruce bog, MN	Yes	Verry and Timmons 1982 Urban and Eisenreich 1988
Thuja peatland, MI	Yes	Kadlec 1983
Forested Swamps and Riparian Zones		
Riverine cypress swamp, SC	Yes	Kitchens et al. 1975
Riparian forest, GA	Yes	Lowrance et al. 1984
Fraxinus lakeside wetland, WI	Yes	Johnston et al. 1984
Swamp forest, LA	Yes	Kemp and Day 1984
Riparian forest, MD	Yes	Peterjohn and Correll 1984
Nyssa swamp, NC	Yes	Brinson et al. 1984
Riparian forest, NC	Yes	Jacobs and Gilliam 1985
Reedy Creek Swamp, FL	Yes	Knight et al. 1987
Riparian buffer, IA	Yes	Schultz et al. 1995

NOTE: Table does not generally include constructed wastewater wetlands, which are generally subjected to much greater concentration of nitrogen and are discussed in later parts of this report.

3.2.2.3 PEATLANDS

There have been few studies of the nitrogen retention capacity of natural bogs and fens, as they generally have no or simple outflows and rely to a great extent on inputs from precipitation (Johnston 1991). While peatlands are generally anaerobic, denitrification has not generally been considered a major pathway for nitrogen loss in these systems, at least in studies in Alaska, Massachusetts, and Minnesota (respectively, Barsdate and Alexander 1975; Hemond 1983; Urban and Eisenreich 1988). Studies by Kadlec and Tilton (1979) and Richardson and Marshall (1986) investigated the role of fens in Michigan in retaining nutrients, with the former study involving the addition of wastewater. A multiple-year study by Kadlec (1983) demonstrated that a peatland in Michigan that received wastewater was consistently a sink for nitrogen (75–81% removal) but began to export phosphorus after several years of phosphorus retention.

3.2.2.4 FORESTED WETLANDS

The functioning of forested wetlands, especially riparian zones, as nutrient sinks has been investigated mostly in the southeastern United States and less-so in the Midwest. Kitchens et al. (1975) found significant reduction in nutrients as the waters passed over the swamp. Kemp and Day (1984) and Peterjohn and Correll (1984) described the fate of nutrients as they are carried into riparian forests by agricultural runoff. The former study found that a Louisiana swamp forest acted primarily as a transformer system, removing inorganic forms of nitrogen and serving as a net source of organic nitrogen, phosphate, and organic phosphorus. The latter study in a riparian Maryland forest described the removal of nitrogen and phosphorus from runoff and ground water as the runoff passed through approximately 50 m of riparian vegetation. Significant reductions of both nutrients from runoff were noted in the study. A similar study of a floodplain forest in Georgia found 14% retention and 61% denitrification of nitrogen (for a total loss of 75% of the incoming nitrogen) and 30% retention of phosphorus (Lowrance et al. 1984). The Maryland and Georgia studies did not consider any river flooding in the calculations of their nutrient budgets. These are among the many studies (e.g., Johnston et al. 1984; Lowrance et al. 1984, 1985, 1995, 1997; Jacobs and Gilliam 1985; Cooper et al. 1987; Cooper and Gilliam 1987) that illustrate the potential for riparian forests for reducing nutrient and sediment loads to streams and rivers.

More recently, research on riparian buffers has been conducted in Iowa and Illinois. Schultz et. al. (1997) have noted nitrate removals in Iowa to be very similar to those measured elsewhere (12 mg-N/L of nitrate in field ground water was reduced to 3 mg-N/L in the buffer). Osborne and Kovacic (1993) installed a wetland grass buffer, rather than a forested buffer, between corn fields and a stream channel in an agricultural watershed in Illinois. They found that their buffers reduced nitrate entry into the stream by 35-45%.

3.2.2.5 CASE STUDIES—NITROGEN RETENTION BY WETLANDS IN THE MIDWEST

Two carefully designed multi-year studies of created wetlands have recently provided an extensive data base on nitrogen retention by newly created wetland basins in the Midwest: the Des Plaines River Wetland Demonstration Project in northeastern Illinois, and the Olentangy River Wetland Research Park in central Ohio. These are two major research wetland sites where extensive multi-year data have been collected to evaluate the retention of low concentrations of nutrients more typical of nonpoint-source pollution than wastewater wetlands.

Des Plaines River Wetlands, Illinois

A number of studies of wetland function were carried out with four full-scale constructed wetlands at the Des Plaines River Wetland Demonstration Project in northeastern Illinois (Figure 3.6; Sanville and Mitsch 1994; Kadlec and Hey 1994; Mitsch et al. 1995). In whole-ecosystem experiments in the early 1990s, hydrologic conditions were varied for high and low flow conditions for entire wetlands (average size = 2.4 ha); the studies were carried out over three years.

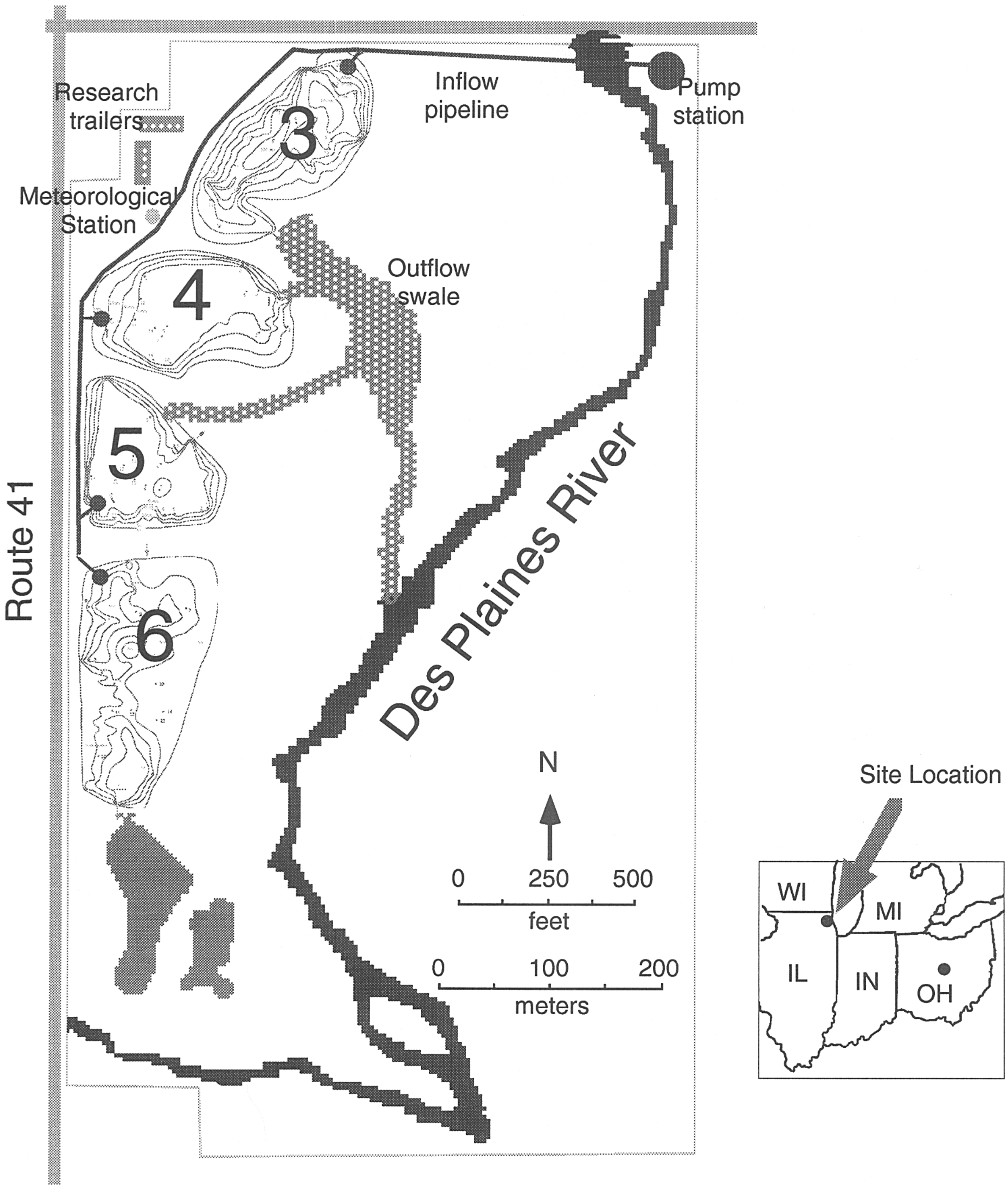


FIGURE 3.6. Four original experimental wetlands at the Des Plaines River Wetland Demonstration Project in northeastern Illinois. NOTE: During the 1989–93 experimental period, experimental wetlands (EW) 3 and 5 were high-flow and 4 and 6 were low-flow.

Researchers had important findings in estimating detention and mixing (Kadlec 1994), water quality function (Hey et al. 1994; Phipps and Crumpton 1994), sedimentation (Fennessy et al. 1994a; Brueske and Barrett 1994), vegetation development (Fennessy et al. 1994b), aquatic metabolism (Cronk and Mitsch 1994a, 1994b), and avian success (Hickman 1994) in created wetlands, most as a function of hydrology.

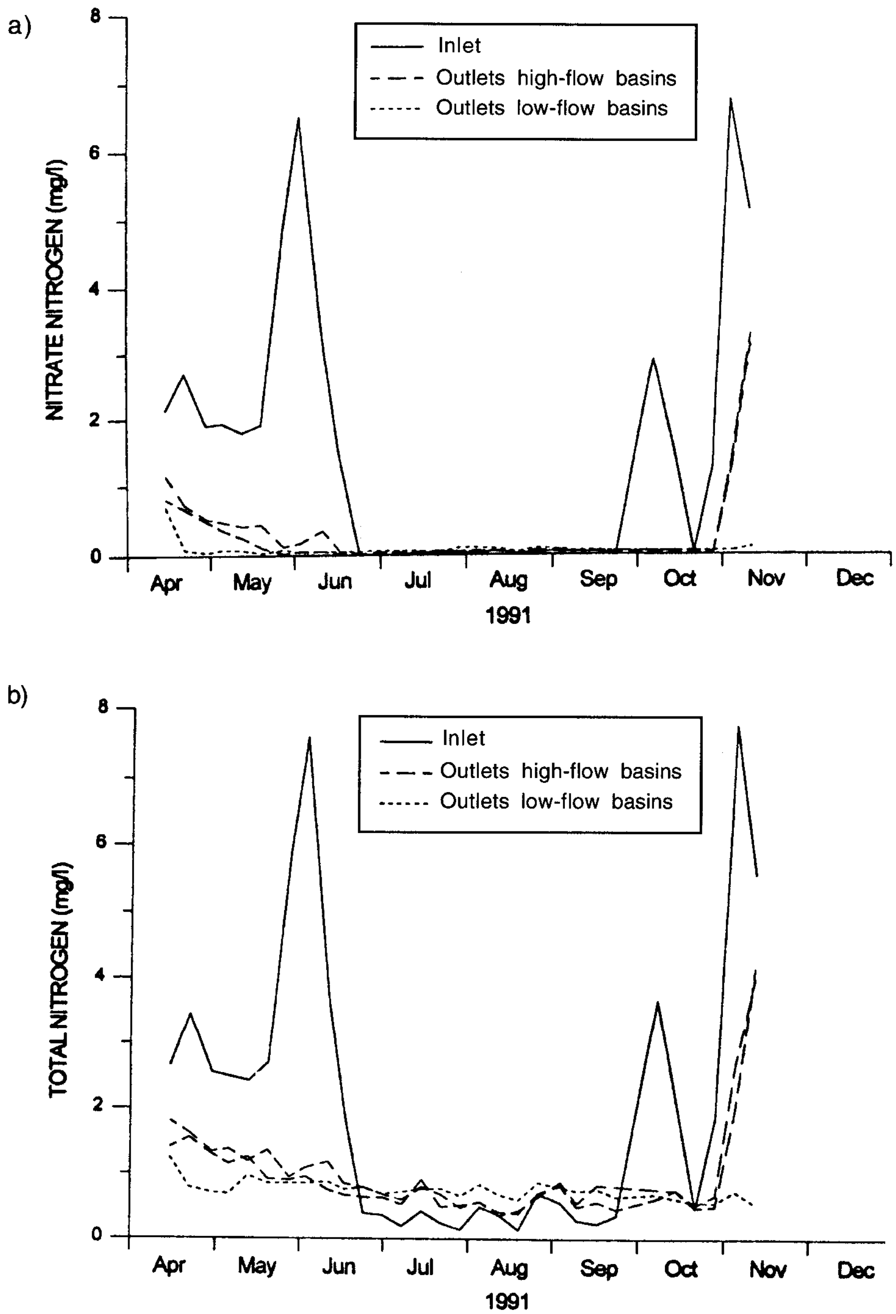


FIGURE 3.7. A strong seasonal pattern of nitrate–nitrogen and total nitrogen typical of midwestern U.S. streams was seen with high concentrations in the spring and fall (inlet data) from the Des Plaines River. NOTE: Outflows are from one low-flow (EW 4) and two high-flow (EW 3 and 5) wetland basins at the Des Plaines Wetland Demonstration Project, shown in Figure 3.6. (*From Phipps and Crumpton 1994.*)

Overall water quality effectiveness of these constructed wetlands for nitrogen are summarized in Tables 3.13 and 3.14 and Figure 3.7. These wetlands received inflow from the adjacent Des Plaines River, with concentrations averaging about 2 mg-N/L, most as nitrate. A strong seasonal pattern typical of midwestern U.S. streams was seen, with very high concentrations in the spring (Figure 3.7), when the wetlands were most effective in retaining nitrogen. Overall, these wetlands reduced influent nitrate–nitrogen by 40–95% over the three years (Table 3.13). For the one year when it was calculated, there was an overall retention of 54–59% of total N, as the wetlands were net sources of organic N (Table 3.14). Overall nitrate–nitrogen retentions were 3–13 g-N m^{-2} yr^{-1} for the low-flow wetland and 11–38 g-N m^{-2} yr^{-1} for the high-flow wetlands.

TABLE 3.13. Nitrate reduction in the experimental wetlands (EW) at the Des Plaines River Demonstration Project, Lake County, IL.

Parameters	**High-Flow Wetlands**				**Low-Flow Wetlands**			
	EW 3		***EW 5***		***EW 4***		***EW 6***	
	In	***Out***	***In***	***Out***	***In***	***Out***	***In***	***Out***
HLR[1] (cm/day)	4.9		4.9		1.5		2.2	
Concentration (*mg-N/L*)								
1989	2.36	1.05	2.32	1.39	2.28	0.11	2.28	0.16
% reduction	55%		40%		95%		93%	
1990	1.87	0.54	1.87	0.53	1.87	0.24	1.87	0.32
% reduction	61%		72%		87%		83%	
1991	1.22	0.23	1.22	0.18	1.22	0.10	1.22	0.18
% reduction	81%		85%		92%		85%	
Avg. % reduction (by concentration)	66%		66%		91%		87%	
Loading (*g-N m^{-2} yr^{-1}*)								
1990	58.1		52.3		15.2		20.4	
1991	14.7		13.4		3.6		3.9	
Retention (*g-N m^{-2}-yr^{-1}*)								
1990	36		38		13		–	
1991	12		11		3		–	

[1]Hydraulic loading rate (HLR) is approximate.
Source: Phipps and Crumpton 1994.

TABLE 3.14. Annual nitrate–nitrogen, organic nitrogen, and total nitrogen budgets for the Des Plaines River experimental wetlands (EW), April–November 1991.

Wetland/ Average Detention Time	**Nitrate**			**Organic N**			**Total N**		
	In	**Out**	**% Loss**	**In**	**Out**	**% Loss**	**In**	**Out**	**% Loss**
High-Flow Wetlands									
EW 3/12 days	21.6	4.7	78%	6.25	8.20	-31%	27.8	12.9	54%
EW 5/13 days	20.2	3.2	84%	6.15	7.48	-22%	26.2	10.7	59%
Low-Flow Wetlands									
EW 4/95 days	3.2	0.2	95%	0.94	0.86	8%	4.1	1.0	75%

NOTE: Nitrogen in and out are in g-N m^{-2} yr^{-1}.
Source: Phipps and Crumpton 1994.

Olentangy River Wetlands, Ohio

The Olentangy River Wetland Research Park at Ohio State University (Figure 3.8) includes two constructed wetland marshes that have been, and continue to be, compared since 1994. One was planted with typical freshwater marsh plants, while the other was left as an unplanted control (Mitsch 1995; Mitsch and Wilson 1996; Mitsch et al. 1998). Inflow nitrate–nitrogen concentrations vary seasonally, as at the Des Plaines River site, and average about 2–4 mg-N/L.

This study area reveals several patterns regarding nitrate–N retention by constructed wetlands (Table 3.15). First, planting vegetation had little effect on nitrate–nitrogen reduction, and differences between the two wetlands were only significant in one year out of three. Second, there is a general decrease in nitrogen retention from the constructed wetlands over the first three years; some is due to the annual differences in hydrology. Third, there is a significant seasonal pattern of higher nitrate–N retention in the growing season, with lower rates of retention in the winter and spring (Figure 3.9). Nevertheless, the annual average nitrate–nitrogen reduction for these wetlands is in a relatively narrow range of 25–28 g-N m^{-2} yr^{-1} over four years, again suggesting this range to be a reasonable starting point for estimating the area required for a given retention of nitrogen in created and restored wetlands in the Midwest.

TABLE 3.15. Average ± standard error (# samples) of weekly nutrient concentrations at Olentangy River experimental wetlands, 1994–96.

Parameters	Inflow	Outflow		% Change, Inflow to Outflow		Results of Paired t-Test Planted vs. Unplanted Outflow p-Value
		Planted Wetland	***Unplanted Wetland***	***Planted Wetland***	***Unplanted Wetland***	
1994—No significant macrophytic vegetation cover in either wetland; heavy algal growth.						
Total Phosphorus *(µg-P/L)*	155±12 (25)	52±8 (25)	42±7 (25)	–66%	–73%	0.030
Soluble Reactive P *(µg-P/L)*	19±7 (23)	4±1 (23)	3±0 (23)	–81%	–84%	ND
NO_3 + NO_2 *(mg-N/L)*	1.71±0.54 (23)	0.87±0.36 (24)	0.92±0.37 (23)	–49%	–46%	ND
1995—Macrophytic vegetation cover is greater in planted than in unplanted wetland.						
Total Phosphorus *(µg-P/L)*	199±21 (34)	82±13 (33)	97±13 (35)	–59%	–51%	ND
Soluble Reactive P *(µg-P/L)*	15±3 (34)	5±1 (35)	9±2 (35)	–65%	–45%	0.037
NO_3 + NO_2 *(mg-N/L)*	1.86±0.25 (35)	1.19±0.22 (35)	1.08±0.21 (35)	–36%	–42%	ND
1996—Both wetlands have approximately the same vegetative cover.						
Total Phosphorus *(µg-P/L)*	191±18 (30)	68±8 (34)	64±9 (35)	–64%	–66%	ND
Soluble Reactive P *(µg-P/L)*	70±11 (29)	8±1 (33)	9±2 (33)	–89%	–87%	ND
NO_3 + NO_2 *(mg-N/L)*	4.42±0.42 (29)	2.97±0.40 (34)	3.30±0.38 (34)	–33%	–25%	0.032

Note: + = increase; – = decrease; ND = no significant difference at $\alpha = 0.05$. Source: Mitsch et al. 1998.

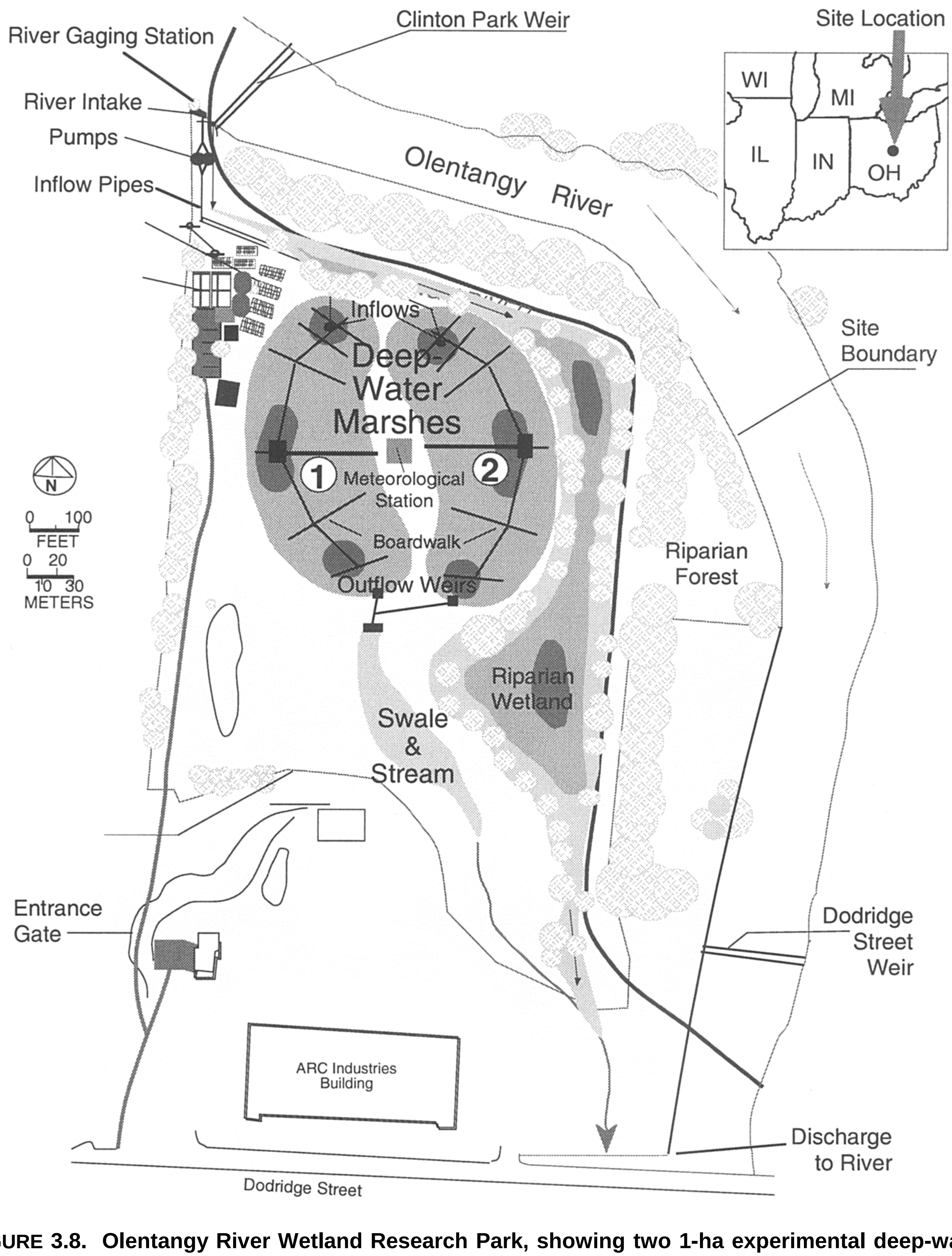


FIGURE 3.8. Olentangy River Wetland Research Park, showing two 1-ha experimental deep-water marshes used in a multi-year study of nitrate–nitrogen retention.

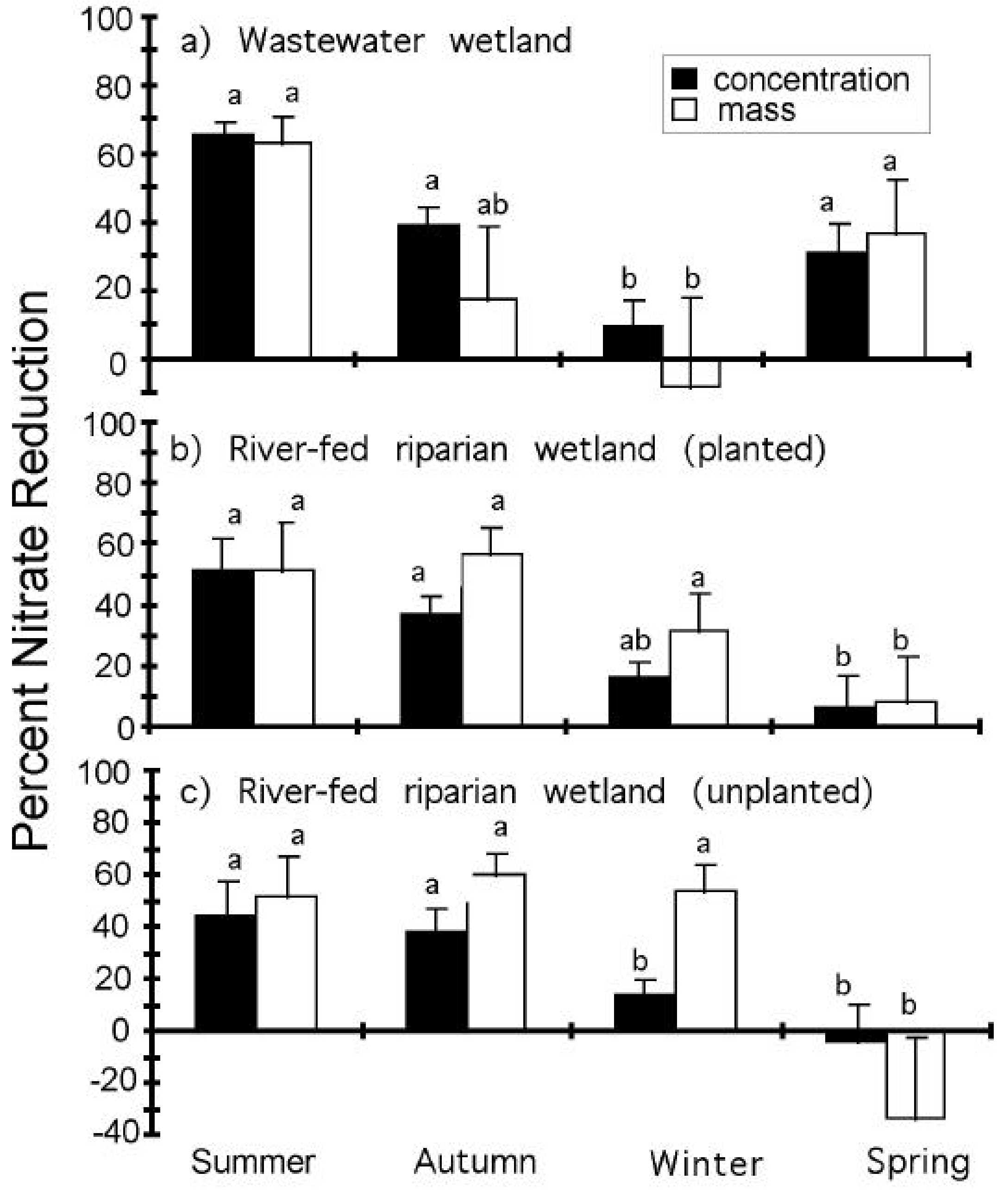


FIGURE 3.9. Seasonal patterns of nitrate retention by mass and concentration during summer, autumn, winter, and spring for (a) wastewater treatment wetland in Licking County, OH, and (b) and (c) 1-ha river-fed created wetlands in Franklin County, OH. NOTE: Date are average ± standard error; same letters indicate no significant differences among all treatments (∝ = 0.05). (*From Spieles and Mitsch 2000.*)

In both of these case studies, investigations of nitrogen retention by constructed midwestern marshes has been one of the aspects investigated over several years. The studies themselves represent a total of 17 wetland-years of study, so general trends should be quite evident for midwestern U.S. regions. The retention of nitrate–nitrogen by concentration for the experimental marshes in Illinois varied between 40% and 95%, while the retention of the Ohio marshes was generally 17–49%. When the wetlands are normalized for flow conditions, there is a generally strong correlation ($r^2 = 0.50$) between percent reduction and flow (Figure 3.10). The multiple wetland years for this study, coupled with the detailed flow and concentration measurements, suggest an achievable ecological engineering design parameter for constructed wetlands for removing nitrate–nitrogen of about 16–24 g-N m^{-2} yr^{-1} (95% confidence interval). This range of retention would thus involve an inflow of about 50 g-N m^{-2} yr^{-1} and would result in a 36–60% retention by mass and about 40–52% retention by concentration (Figure 3.10).

3.2.2.6 DESIGN CONSIDERATIONS AND LIMITATIONS

Both kinds of wetlands, natural and constructed, have been shown to be effective sinks for nutrients, especially when the nutrient loads are not excessive. But each type has its benefits and shortcomings. With fewer and fewer natural wetlands due to drainage and land conversion, many natural resource managers and policymakers believe that we should not be adding any type of strong pollution to our remaining natural wetlands (Olson 1992). Constructing wetlands for controlling nonpoint-source pollution is a good alternative, even though there were some early indications that we have not been building wetlands correctly (Erwin 1991; Mitsch and Wilson 1996), especially when they have been built to replace the function of a wetland lost for some type of human development project. Wetlands need to be designed, constructed, and restored in an ecologically sound and predictable manner (Mitsch and Cronk 1992; Mitsch 1992; Mitsch and Wilson 1996). Some of the many important variables to consider when creating and restoring wetlands for controlling nonpoint-source pollution are discussed here.

Loading Rates

Loading rates (flow times concentration of inflowing water) dictate the effectiveness of wetlands in reducing nitrogen (Figures 3.10 and 3.11). Empirical models, such as these graphs, can be used as first estimates of the potential nutrient retention in freshwater wetlands. Extensive experience with flow-through wetlands (see case studies above) suggested a narrow range that centers around 16–24 g-N m^{-2} yr^{-1} as a reasonable target for wetland retention of nitrate–nitrogen. The loading rate, assuming 50% retention, therefore, would be about 32–48 g-N m^{-2} yr^{-1}. This is the equivalent of 10 m/yr of inflow to a wetland with a concentration of 3–5 mg-N/L. This flow is approximately 10 times normal rainfall for at least the eastern half of the MRB. If flows are less, concentrations would have to be greater to effectively load the wetland.

Inflows to wetlands in rural areas are often due to pulses of runoff, stream flow, and/or river flooding. Storm events, if significant enough, can cause nitrogen to shoot through the system, bypassing effective retention (Figure 3.12). The optimum detention time has been suggested to be 5–14 days for treating municipal wastewater. Brown (1987) suggested a retention time of a riparian wetland system in Florida of 21 days in the dry season and more than 7 days in the wet season.

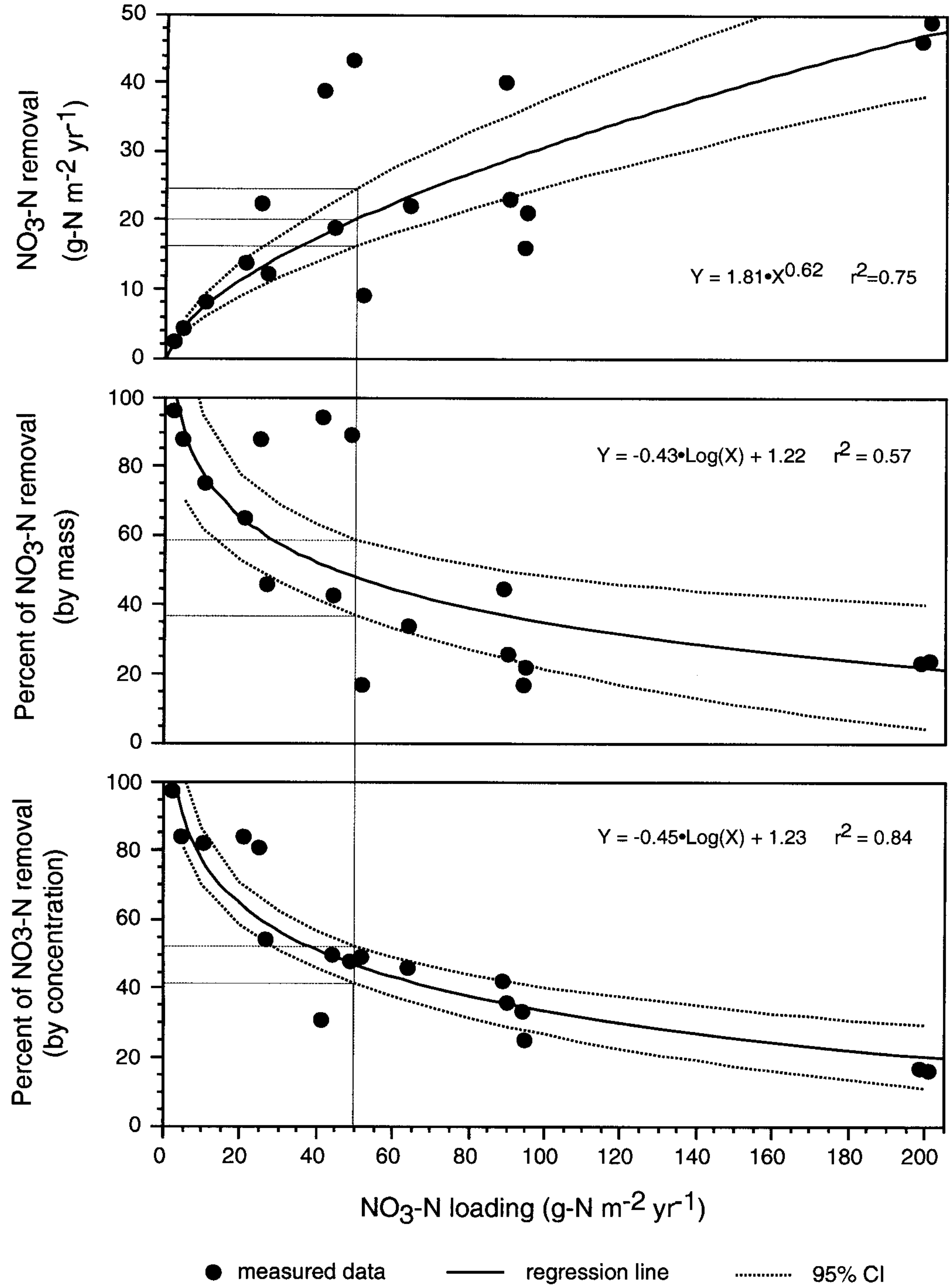


FIGURE 3.10. Summary of 17 wetland-years of nitrate–nitrogen retention data from the Des Plaines River and Olentangy River experimental wetlands. NOTE: Wetland removal, percent removal by mass, and percent removal by concentration are plotted versus nitrate–nitrogen loading; 95th confidence intervals (CI) are also shown. Each data point represents one wetland-year.

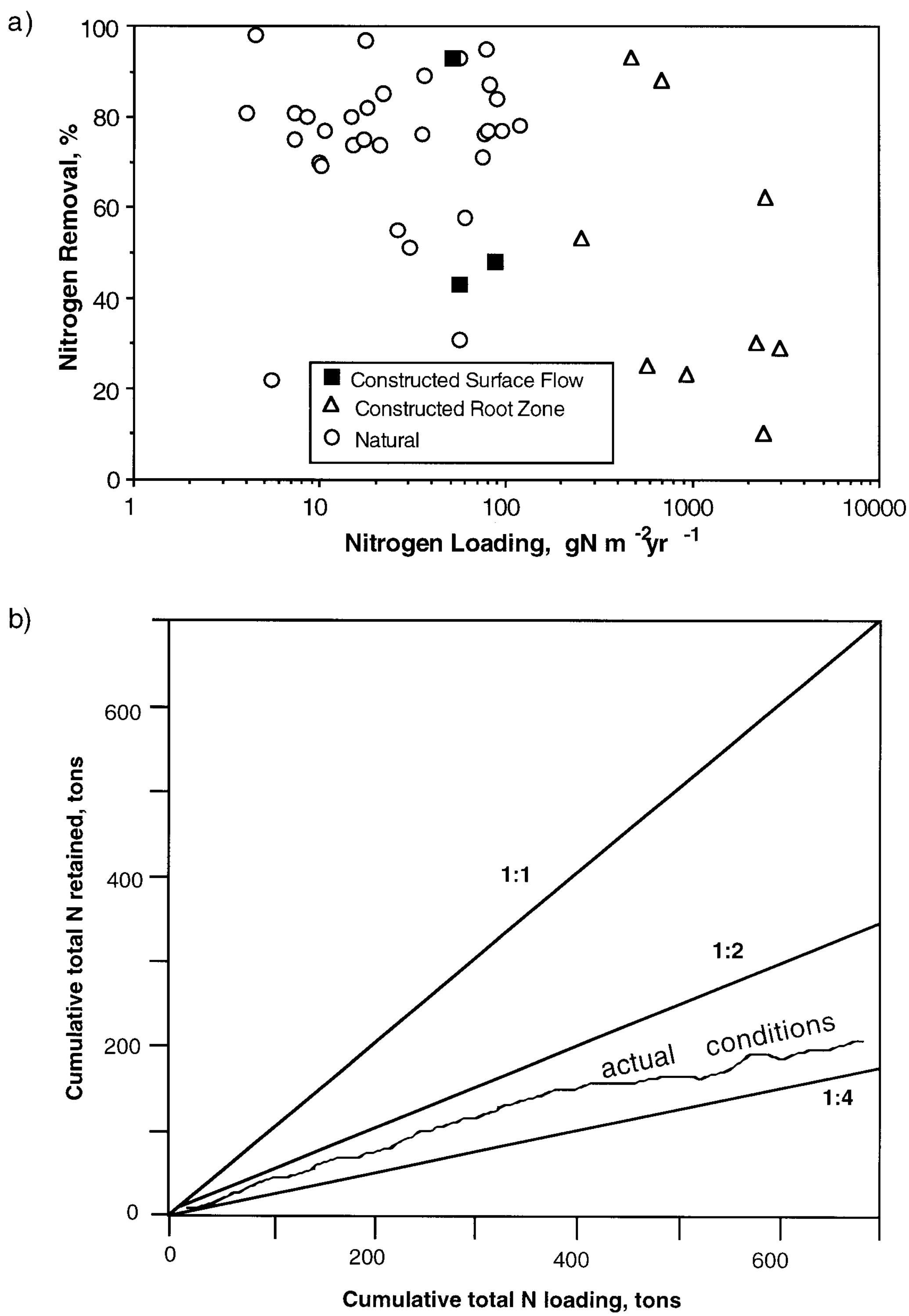


FIGURE 3.11. Examples of wetland nitrogen retention versus loading rate: (a) nitrogen retention from several constructed and natural wetlands (*from Mitsch and Gosselink 1993*)**, and (b) cumulative nitrogen mass retained versus mass loading of a marsh receiving low-level nutrients for eight years.** (*From Moustafa et al. 1996*).

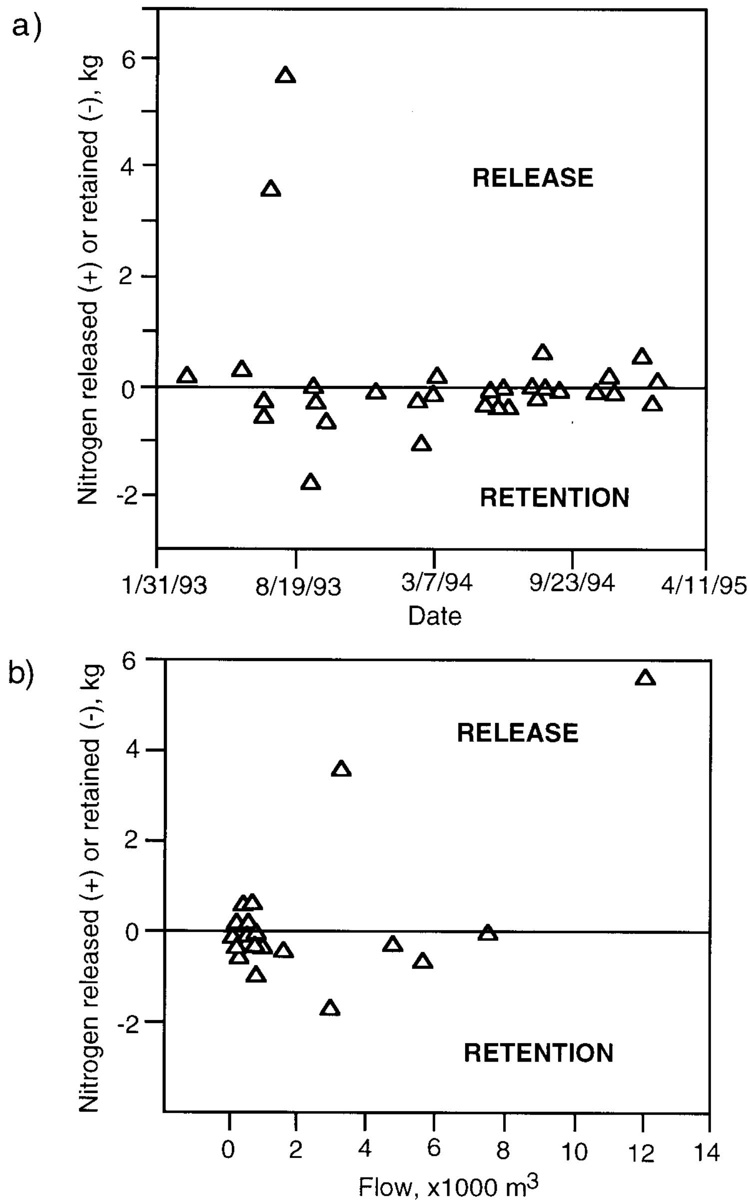


FIGURE 3.12. Nitrogen retention in an Australian wetland in an agricultural area during storm events showing (a) seasonal pattern of retention and release, and (b) effect of storm flow on retention or release. (*From Raisin et al. 1997.*)

Soil Carbon

The organic carbon content of soils has great significance for many processes in a wetland, particularly denitrification. Microbes that carry out the process of denitrification require an organic carbon source for energy (Mitsch and Gosselink 1993). Furthermore the organic carbon also creates the sufficiently low redox conditions necessary for denitrification to occur. Wetland soils generally vary between 5% and 75% organic matter, with higher concentrations in peat-building systems, such as bogs and fens, and lower concentrations in mineral soil marshes subject to mineral sedimentation or erosion or in newly constructed wetlands. Riparian forests must have low concentrations of organic matter, about 5%, relative to wetlands. When wetlands are created or restored on mineral soils, one of the factors that most limits their being equivalent to natural wetlands is the accumulation of soil organic carbon (Mitsch and Flanagan 1997). This accumulation takes many years on drained or upland soils that have been converted to wetlands.

Temperature/Seasonal Affects

There is a definite effect of temperature and, hence, length of growing season, on biological processes such as denitrification. Studies of wastewater wetlands and nonpoint-source wetlands in central Ohio (Spieles and Mitsch 2000) show that percent removal of nitrate–nitrogen is related to season, with considerably less nitrate–nitrogen retention in nongrowing season months compared to growing season months (Figure 3.9).

Landscape Position

Wetlands should not be expected to control all of the influx of nutrients from a watershed, nor should one small wetland be expected to result in significant improvements in downstream water quality. If wetlands are to be constructed in the watershed for controlling nonpoint-source pollution, there are many possible positions in the landscape (Figures 3.13 and 3.14), including instream wetlands, riparian wetlands, and terraced wetlands.

Wetlands can be designed as instream systems by adding control structures to the streams themselves, or by impounding a distributary of the stream (Figure 3.13a). Blocking an entire stream is a reasonable alternative only in low-order streams (see also Section 3.2.4, Controlled Drainage). This design is particularly vulnerable during flooding and may be very unpredictable in its ultimate stability. It has the advantage of potentially treating a significant portion of the water that passes that point in the stream. Maintenance of the control structure and the distributary may require significant management commitments to this design. The natural design for a riparian wetland fed primarily by a flooding stream (Figure 3.13b) allows for river floods to seasonally deposit sediments and chemicals in the wetland (see Section 3.2.3, Riparian Buffers). Because both man-made and natural levees are along major sections of streams, it is often possible to create such a wetland with minimal construction work. The wetland could be designed to capture flooding water and sediments and slowly release the water back to the river after the flood passes. This is the design of natural riparian wetlands in bottomland hardwood forest areas. The wetland could also be designed to receive water from flooding and retain it by using flap-gates.

Multiple wetlands can be constructed in the landscape to intercept small streams and drainage tiles (Figure 3.13c, d). The main stream itself is not diverted, but the wetlands receive their water, sediments, and nutrients from small tributaries, swales, and overland flow. More significant, if tile drains can be located and broken upstream of their discharge into tributaries, they can be very effective conduits for supplying adequate water to the wetlands. Because these tile drains are often the sources of the highest concentrations of chemicals, such as nitrates, the wetlands could be very effective in controlling nonpoint-source pollution when located downstream of them.

The advantages of locating several small wetlands in the upper reaches of a watershed (but not in the streams themselves), rather than fewer larger wetlands in the lower reaches, should be considered (Figure 3.13e). Loucks (1990) argues that a better strategy for wetlands to survive extreme events is to locate a greater number of low-cost wetlands in the upper reaches of a watershed, rather than building fewer high-cost wetlands in the lower reaches. A modeling effort on flood control by Ogawa and Male (1983) suggested the opposite: the usefulness of wetlands in decreasing flooding increases with the distance the wetland is downstream.

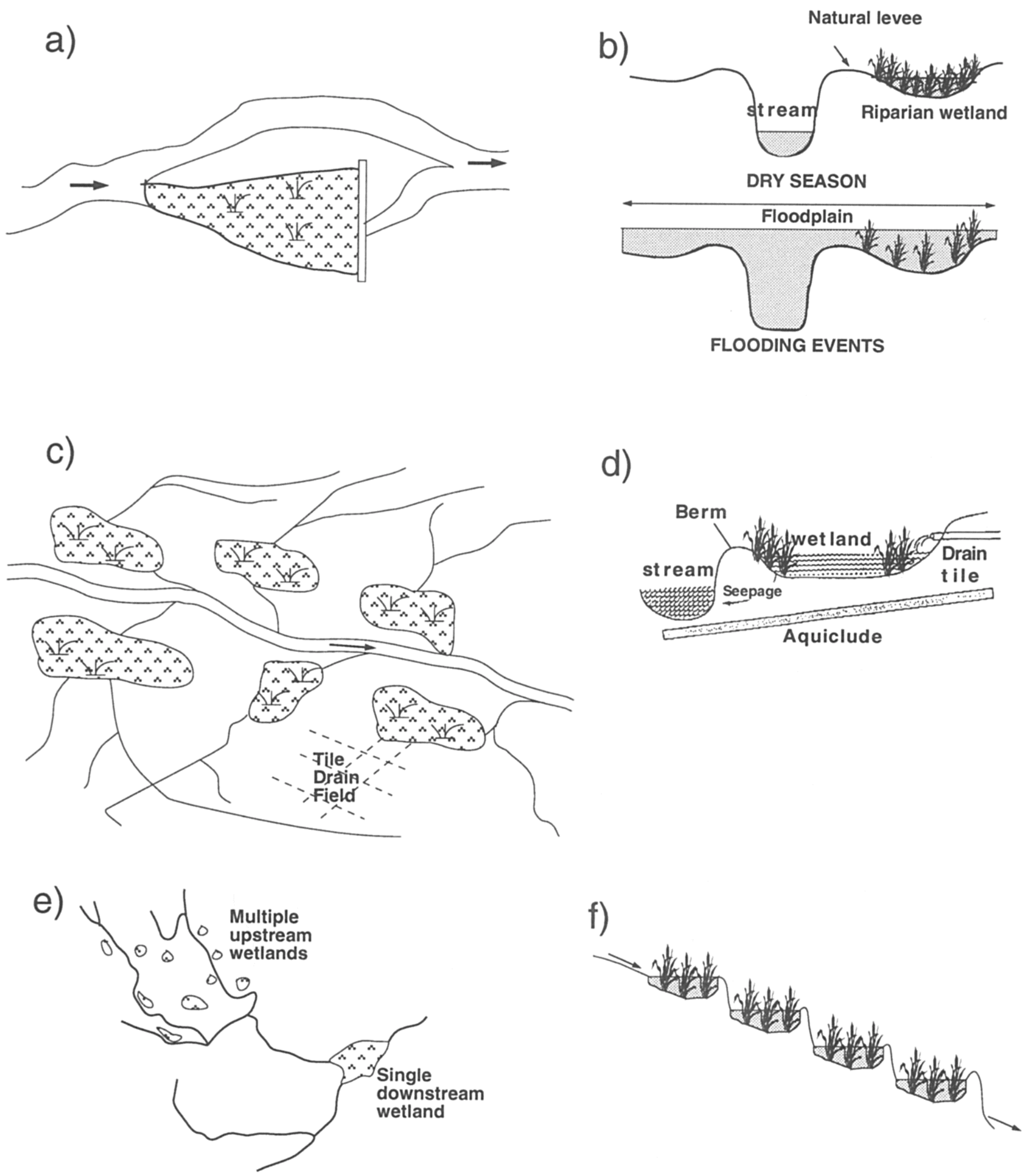


FIGURE 3.13. Examples of locations of created/restored wetlands in the landscape: (a) riparian bifurcation of river; (b) riparian wetland with seasonal flooding; (c) created marshes in small streams and intercepting tile drainage; (d) details of lateral wetland intercepting ground water carried by drainage tiles; (e) landscape location choices for wetlands; and (f) terraced in hilly terrain. (*Illustrations from Mitsch 1992, Mitsch and Cronk 1992, Mitsch and Gosselink 1993, and Kovacic et al. 1995*.)

Wetlands are a phenomenon of naturally flat terrain. However, steeper terrain is often most susceptible to high erosion and, hence, high contributions of suspended sediments and organic nitrogen. One approach is to attempt to integrate terraced wetlands into the landscape (Figure 3.13f). In this case, wetland basins are constructed as smaller basins that stair-step down steep terrain. While there are some examples of these types of wetlands, particularly in the building of acid mine drainage wetlands in the Appalachians, few wetlands have been constructed of this type for controlling nonpoint-source pollution.

Figure 3.14 illustrates a hypothetical small watershed with a combination of livestock runoff treatment wetlands, nutrient/sediment wetlands, grassed waterways, and riparian buffers. All would be effective in reducing nonpoint-source pollution in the agricultural landscape.

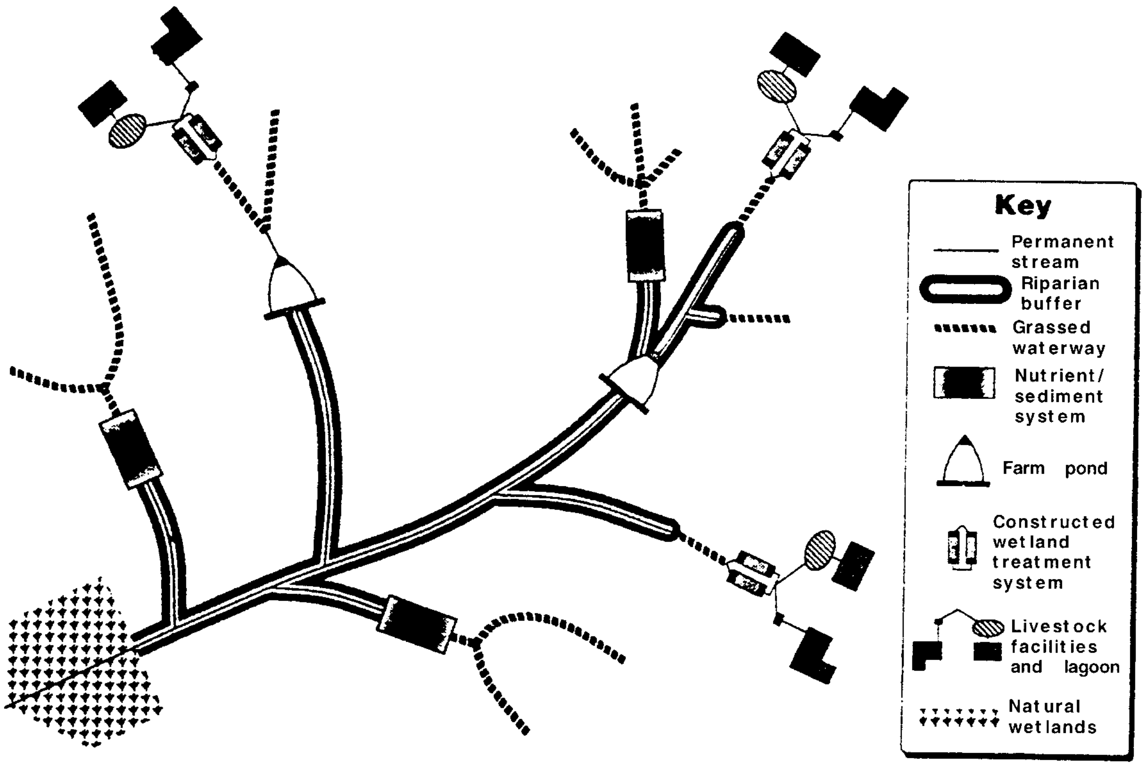


FIGURE 3.14. Hypothetical locations of riparian buffers, farm ponds, constructed wetlands, and grassed waterways in region of livestock facilities. (*From Hammer 1992.*)

3.2.3 Riparian Buffers

Riparian buffers (Figure 3.15) are vegetated areas next to water resources that provide protection from nonpoint-source pollution and provide bank stabilization and aquatic and wildlife habitat. The formal definition of riparian buffer is diverse and depends on the individual or group defining the term. The USDA Forest Service (Welsch 1991) defines a riparian buffer as:

> "the aquatic ecosystem and the portions of the adjacent terrestrial ecosystem that directly affect or are affected by the aquatic environment."

This includes streams, rivers, lakes, and bays and their adjacent side channels, flood plain, and wetlands. In specific cases, the riparian buffer may also include a portion of the hill slope that directly serves as streamside habitats for wildlife. Leading experts (Lowrance et al. 1985) on riparian buffers define them as follows:

> "a complex assemblage of plants and other organisms in an environment adjacent to water."

Without definitive boundaries, riparian buffers may include stream banks, floodplain, and wetlands, as well as sub-irrigated sites forming a transitional zone between upland and aquatic habitat. Mainly linear in shape and extent, they are characterized by laterally flowing water that rises and falls at least once within a growing season. Natural riparian buffers are composed of grasses, trees, or both types of vegetation. If riparian buffers are maintained or reestablished, they can exist under most land uses: natural, agricultural, forested, suburban, and urban.

Since most existing or planned riparian buffers are forested, this discussion focuses on riparian forest systems. The ideal forested riparian buffer system (Figure 3.15a) was originally described by Welsch (1991) and has generally been accepted as a model for riparian buffers (Lowrance et. al. 1997). Certainly, variations of the ideal buffer can be used to satisfy local conditions, as described later, but the mechanisms of pollutant removal must be understood before any system can be designed and used to its maximum capability. Integrated streamside riparian buffers (forest and grass or shrub) that are designed to intercept surface runoff and subsurface flow can be effective in controlling nonpoint-source pollution by removing nutrients, especially nitrogen and sediment (USDA 1997).

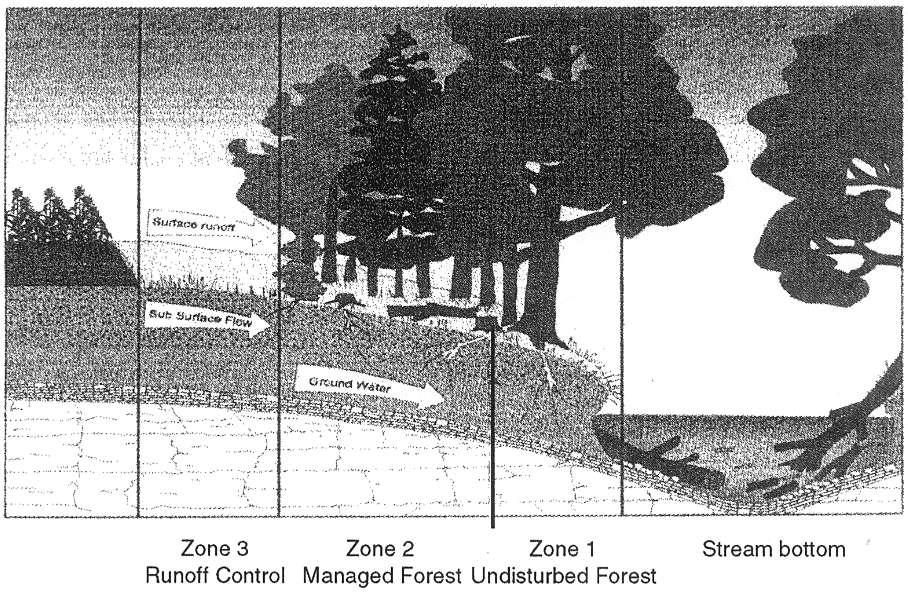
Many factors determine the effectiveness of riparian buffers for any given pollutant. Hydrology is the most important of these factors (Hill 1996). For example, removal of contaminants from surface runoff requires that runoff water be sufficiently slowed to allow sediment to settle out. If the runoff water does not spread over the buffer, it will move through the buffer in channels. Channelized water moves almost as quickly through a buffer as it does from the field, thereby making the buffer ineffective at pollutant removal (Dillaha et al. 1989). This is the primary function of zone 3, shown in Figure 3.15a. Most nitrogen from agricultural fields reaches surface water as nitrate in the ground water below the soil surface. For nitrate to be removed from ground water before it reaches surface water, the ground water must enter a zone where plant roots are or have been active. These plant roots may absorb the nitrate for use in plant growth or, more important, may provide an energy source for bacteria that converts nitrogen in nitrate to a gas, which then escapes to the atmosphere. This process, denitrification, occurs almost exclusively in water-saturated zones, where abundant organic matter is present. Zones 2 and 3 in Figure 3.15a are primarily for removal of nitrate from subsurface flows, although they may serve other very important purposes that enhance overall stream health.

The large majority of riparian buffer sites that have been investigated for nitrate removals have shown that nitrate concentrations in shallow ground water were significantly reduced as the water flowed through the riparian buffer (Gilliam et al. 1997a, 1997b). However, it is possible for nitrate to pass below the riparian buffer at depths far enough below the root zone where very little nitrate removal occurs (Correll et al. 1994). It is also possible for ground water to move through the riparian buffer so quickly that removal is limited (Haycock and Pinay 1993). This latter case would occur when subsurface drainage tiles are present.

3.2.3.1 DESIGN CONSIDERATIONS AND LIMITATIONS

Scientists agree that a corridor of vegetation can be effective at buffering valuable aquatic resources from the potential negative impacts of human use of the adjacent land. The streamside vegetated buffer filters nonpoint-source pollutants from incoming runoff (see Table 3.12) and provides habitat for a balanced, integrated, and adaptive community of riparian and aquatic organisms (Welsch 1991). These filtering and habitat functions are often best provided by natural vegetation ,such as trees and associated woodland or forest plants in the zone directly adjacent to the waterway. While there is general agreement about the benefits of buffers, the specific design criteria, such as buffer width, types of vegetation, and management, are the subject of considerable debate.

a)



b)

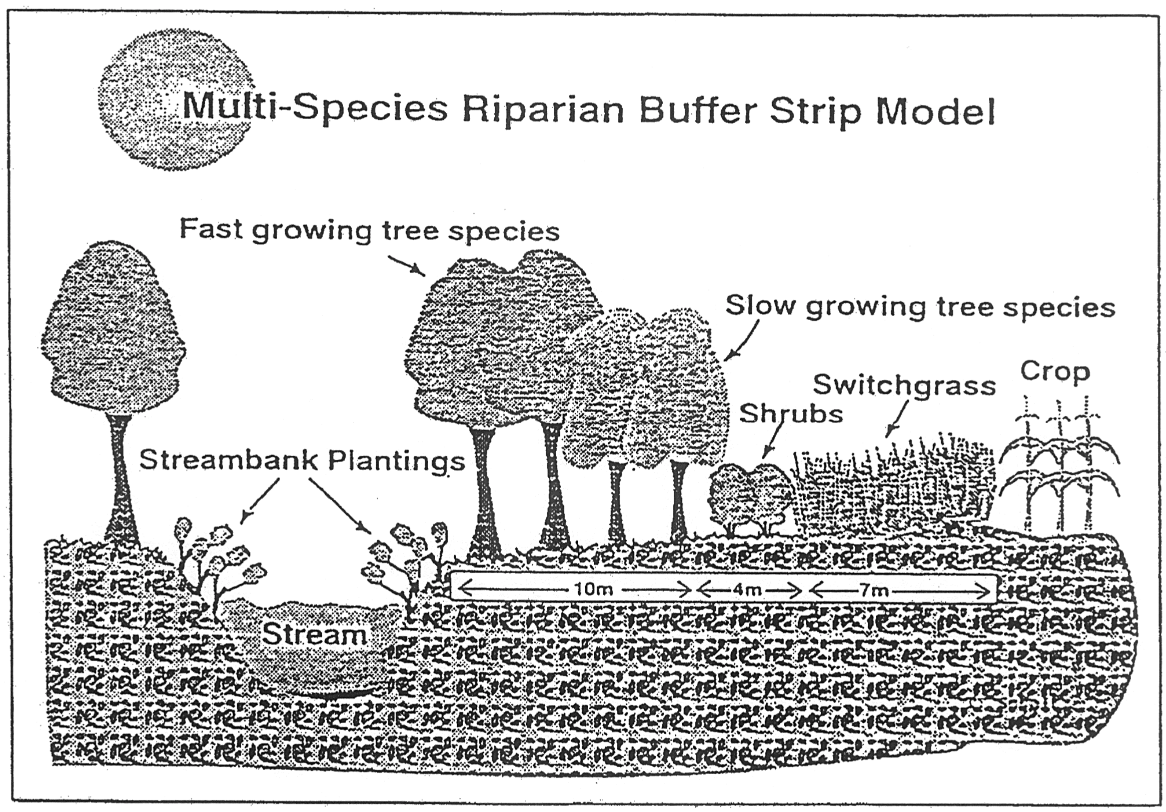


FIGURE 3.15. Schematics of riparian buffer zones: (a) three-zone riparian buffer system (*from Welsch 1991*)**, and (b) multi-species riparian buffer strip model which includes tree rows closest to the stream, shrubs, and a strip of switchgrass adjacent to the cropland.** (*From R. Schultz.*)

Buffer Width

Width is considered the most important controllable variable in determining the effectiveness of buffers in reducing pollutants and protecting stream health. Buffers that are too narrow may not be sustainable or effective in protecting stream banks. Conversely, buffers that are wider than needed limit the use of adjacent land and are unpopular with landowners. Complicating the determination of design buffer widths are the effects of varying site characteristics associated with topography, hydrology, geology, and land use. Additionally, other factors, such as the value of the water resource and adjacent land, must be considered when determining widths.

The width of most existing riparian forest buffers was established by leaving the area adjacent to the stream as forest. This area was generally too wet or too steep to be used conveniently for agricultural or urban purposes. Welsch (1991) recommended a widely acclaimed riparian buffer system that was 29-m wide on both sides of the stream. There is little debate among riparian buffer experts that the system he described is very good as an idealized multipurpose buffer to protect all aspects of stream quality. However, requiring this width along every stream is probably not necessary to protect the streams from nonpoint-source pollution and will result in widespread opposition from landowners. The width necessarily depends upon what functions are expected of the riparian buffer and the site characteristics.

Most decisions about buffer widths will be a compromise between ideal widths based on environmental goals (wildlife corridors, bank stabilization, water quality protection) and sociological or economic constraints. Science-based criteria, for which research data may be available to support an informed decision, include the functional value of the water resource; watershed, site, and buffer characteristics; adjacent land use; and buffer function. The functional value of the water resource is important for determining buffer width in that a highly valued resource may merit a wider buffer for increased protection.

Buffer Zonation

Watershed, site, and buffer characteristics are most important when evaluating pollutant-filtering effectiveness. The size and topography of the watershed determine the amount and rate of surface and ground water passing through the buffer. Site characteristics, such as soil type, slope steepness, microbial populations, and vegetation, determine the amount of pollutants that are filtered out of the water before it enters the waterway. Buffer characteristics, such as the types of vegetation and their location in the buffer, can also influence pollutant removal effectiveness.

The most widely recognized buffer planning model is the three-zone buffer developed by the USDA Forest Service (Welsch 1991). Zone 1 of the model begins at the normal water level or at the edge of the active channel and extends a minimum of 4.5 m along a line perpendicular to the water course. Dominant vegetation consists of existing or planted woody vegetation suitable for the site and intended purpose. This zone should remain undisturbed; therefore, tree removal is generally not permitted. Zone 2 begins at the edge of zone 1 and extends a minimum of 18 m perpendicular to the water course. While vegetation in zone 2 should be similar to that of zone 1, removal of trees and shrubs is permitted, provided they are replaced. The zone 3 begins at the outer edge of zone 2 and has a minimum width of 6 m. Vegetation in this zone can be grazed or ungrazed grass or other plant communities, as long as it facilitates sediment filtering, nutrient uptake, and the conversion of concentrated flow to uniform, shallow, sheet flow through the use of structural practices, such as level spreaders (Lowrance et al. 1995).

Most nitrogen from nonpoint sources enters surface waters as nitrate–nitrogen in ground water. As the shallow ground water moves through the riparian buffer, microorganisms change the nitrate–nitrogen to gaseous nitrogen via a process known as denitrification. When the soil is poorly aerated (anaerobic conditions), some microorganisms reduce nitrates to the gaseous components of nitrous oxide, nitric oxide, or free nitrogen gas.

Denitrification is most effective in root-zone soil layers, where carbon sources are available for the denitrifying bacteria. Numerous researchers have reported that it is the complex interaction between vegetation and below-ground environment that provides the appropriate conditions for denitrification to occur (Lowrance et al. 1995). The area of interaction within the riparian buffer is generally quite narrow—3–15 m—from the field through the riparian buffer. The majority of denitrification that has been observed in riparian buffers occurred within the first 3 m of the forested riparian buffer. Denitrification as measured in coastal plain forested riparian buffer areas has removed as much as 29 g N m^{-2} yr^{-1}. Typically, though, denitrification rates are generally 2–6 g N m^{-2} yr^{-1} (Table 3.11).

Most studies indicate that denitrification takes place throughout the year (Lowrance et al., 1995). Climate would certainly have an influence on the amount of denitrification taking place during the winter months but it must be remembered that the primary processes are occurring in subsoils where temperatures are much higher than average winter air temperatures.

Vegetation

Vegetation in riparian buffers also removes nutrients through uptake. Some of these nutrients are sequestered in woody vegetation, whereas the nutrients absorbed into herbaceous materials generally are recycled as the vegetative matter dies. Several studies have indicated that uptake by above-ground, woody vegetation removes various amounts of nitrogen and phosphorus, depending on the riparian conditions (Table 3.16). Although nitrogen uptake by the vegetative portion of the riparian buffer contributes to nitrogen reductions, denitrification is the primary process that removes nitrate from the shallow ground water that flows through riparian buffers.

TABLE 3.16. Above-ground woody vegetation uptake of nitrogen and phosphorus in coastal plain riparian forests.

References	Rivers/Locations	Nitrogen ($g\ m^{-2}\ yr^{-1}$)		Phosphorus	
		Total Input	*Woody Storage*	*Total Input*	*Woody Storage*
Correll & Weller 1989	Rhode R., MD	ND[1]	1.2–2.0	ND[1]	0.3–0.5
Peterjohn & Correll 1984	Rhode R., MD	7.7	1.2	1	0.17
Fail et al. 1986, 1987 (mean)	Little R., GA	11.4	5.2	0.75	0.38
Fail et al. 1986 (max.)	Little R., GA	19.44	9.76	1.26	0.69
Fail et al. 1986 (min.)	Little R., GA	8.0	3.46		0.19

[1]*ND = not determined.*
Source: Lowrance et al. 1995.

3.2.3.2 COMPARISON OF WETLANDS AND RIPARIAN BUFFERS

Scientific data thus far obtained show that wetlands and riparian buffers are not equally effective in all situations for nitrate reduction. This was clearly recognized by Lowrance et al. (1997) for the various physiographic provinces present in the Chesapeake Bay watershed. The scientists assembled by Lowrance spent a large amount of time applying available riparian buffer information to the landscape conditions present in the watershed in an attempt to estimate how effective buffers would be for nitrate removal. They concluded that in areas where the excess precipitation moves across, in, or near the root zone, riparian buffers should retain 50–90% of the nitrogen entering the buffer. Lower removals by buffers would be expected where these conditions do not exist. For example, where ground-water flow moves so far below the root zone of riparian buffers that little nitrate uptake can occur or little organic matter from the roots gets into the groundwater, little or no loss would be expected.

Drainage water that enters a drainage line and is piped to a collector ditch or stream may be more suitable for a wetland basin than a riparian zone. This situation is very common in many agricultural areas of the Mississippi River Basin. Another common situation is the shallow ground water is intercepted by a shallow field ditch and transported through the drainage system. In either of these conditions, riparian buffers are not an attractive alternative. At sites in Great Britain, buffers were planted beside streams that had drainage tubes from agricultural fields entering into them. Just as one would predict, the buffers had little or no effect on nitrate entering the stream.

The limitations with removing nitrate from tile drainage water with only riparian buffers was recognized by the scientists in both Illinois (Kovacic et al. 1995) and Iowa (Schultz et al. 1995) in their experimental system (Figure 3.15b). Both groups used constructed wetlands in addition to streamside buffers to treat tile drainage waters. Kovacic (personal communication) estimates that they are currently removing 46% of the nitrate from tile drainage water entering the wetland. He also estimates that 1–3% of the land currently under cultivation would be required for wetlands in areas similar to those in their study to achieve this level of nitrate reduction. There is little question that small wetlands can be used to make a significant reduction in the nitrate entering streams from agricultural lands, but construction and placement of these treatment areas will be extremely critical to both their effectiveness and their acceptance by landowners.

A very similar situation to interception of ground water is interception by an open field ditch. This ditch may be an old stream that has been channelized or an original drainage way. In either situation, much less nitrate is removed from the drainage water as it moves into the surface water, compared to natural streams with riparian buffers. In many places where this situation exists, it may not be practical to have buffers along the ditch and still be able to farm the land. The only apparent option in these situations for nitrate removal is either wetlands or controlled drainage.

3.2.4 Controlled Drainage

Improved drainage has increased crop production in much of the Mississippi River Basin. However, there is a potential for managing drainage systems in ditches and small streams to satisfy agricultural production needs and at the same time minimize adverse environmental effects. This approach is called controlled drainage. Currently, there is very little controlled drainage in the Mississippi River Basin, although this is an accepted best management practice in several states along the Atlantic Coast. Controlled drainage is very popular in North Carolina with both farmers and environmental regulators because the practice is responsible for increased yields and reduced nutrient losses to surface water.

3.2.4.1 DESIGN CONSIDERATIONS AND LIMITATIONS

Surface versus Subsurface Drainage

Drainage is accomplished by two methods: (1) open-ditch systems designed to provide primarily surface drainage (surface runoff) or (2) underground systems comprised of drain tile or tubing designed to lower the water table by subsurface flow. Subsurface drainage is obtained by buried tile or tubing (10–15 cm diameter) that is placed 1–2 m deep and 7–70 m apart. A subsurface system provides drainage when the water table rises above the drain depth and water flows toward and into the drain. The drainage process whereby water infiltrates into the soil and moves within the soil profile is referred to as subsurface drainage, shallow-ground water flow, or sometimes interflow.

In practice, it is often difficult to differentiate between surface and subsurface drainage, because the outflow in drainage ditches or canals is usually a combination of both surface and subsurface flow. The relative proportion of surface and subsurface flow in the total drainage volume depends on many factors, including rainfall intensity, land surface roughness and slope, vegetation, soil permeability, and ditch or drain tubing spacing and depth. Open ditches are normally spaced farther apart than buried tubing, which typically causes subsurface flow to be slow, resulting in collection of predominately surface drainage. But in highly permeable soils, open ditches may provide significant subsurface drainage.

The difference in drainage method (surface versus subsurface flow) is important from a water quality standpoint because the characteristics of the two drainage waters differ. Surface drainage systems result in rapid removal of excess water over a relatively short time period. This water flowing over the land surface has relatively high energy sufficient to detach and transport soil particles and constituents attached to them, such as phosphorus, organic nitrogen, and many pesticides (Gilliam et al. 1978; Skaggs and Gilliam 1981; Deal et al. 1986). Subsurface drainage typically contains very little sediment and high concentrations of soluble constituents, such as nitrate–nitrogen (Gilliam et al. 1978; Logan et al. 1980; Skaggs and Gilliam 1981; Skaggs et al. 1982; Evans et al. 1987; Deal et al. 1986; Randall and Vetch 1995; Randall et al. 1997).

Water Control Structures

Water-control structures, such as a flashboard riser, installed in the drainage outlet allow the water in the outlet to be raised or lowered as needed. This water management practice has become known as controlled drainage. When the flashboards are lowered or removed, subsurface drainage occurs more quickly (Figure 3.16). When flashboards are added to the riser, the subsurface drainage rate is decreased, and the height of the water level in the ditches and surrounding fields rises. Using controlled drainage to manage the field water allows timely drainage, but also maximum storage of water within the field for crop use.

The transport of nitrogen from drained fields can be minimized by managing the drainage system such that only the minimum drainage water necessary is allowed to exit the field. In numerous field studies in North Carolina (Gilliam et al. 1978, 1979; Skaggs et al. 1982; Deal et al. 1986; Evans et al. 1989), drainage control reduced the annual transport of total nitrogen at the field edge by 45% on average. Working in Ontario, Canada, Drury et al. (1996) obtained results almost identical to those in North Carolina. The crop, soil and climatic conditions in Canada should be very similar to those in parts of the northern section of the Mississippi watersheds.

Nitrogen reductions from controlled drainage result from two processes. First, controlled drainage reduces the volume of drainage water leaving a field by 20–30% on average; however, outflow varies widely, depending on soil type, rainfall, type of drainage system, and management intensity. During dry years, controlled drainage may totally eliminate outflow. In wet years, control may have little or no effect on total outflow. Second, controlled drainage provides a higher field water table level, which promotes denitrification within the soil profile. In some cases, nitrate–nitrogen concentrations have been 10–20% lower in outflow from controlled systems, compared to uncontrolled, free-draining systems (Evans et al. 1989). The combined effect of reduced flow and reduced nitrate concentration results in the overall 45% reduction in nitrogen mass transport at the field edge (Figure 3.17). Controlled drainage has also been documented to reduce phosphorus transport by roughly 35%.

a)

b)

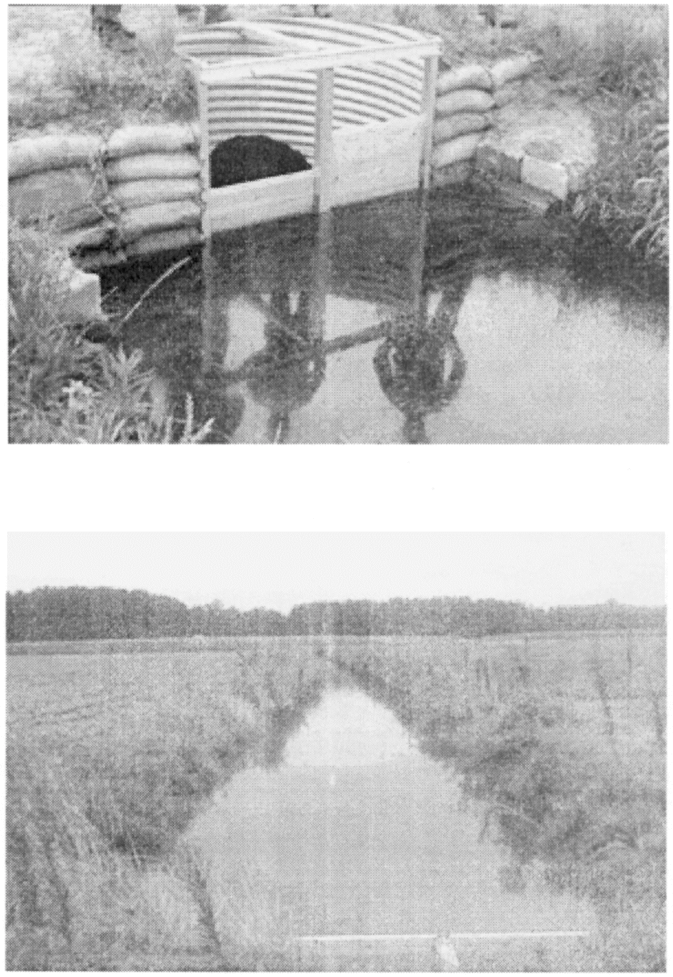


FIGURE 3.16. Controlled drainage system showing (a) flashboard riser and (b) water profile in drainage ditch upstream of flashboard riser.

Topography

Successful management of controlled drainage systems rests on two important objectives: (1) achieving optimum production efficiency and maximum nutrient utilization by the crop; and (2) attaining maximum water quality benefits. Controlled drainage structures require that the topography be relatively flat. The costs of installing and maintaining them will usually exceed their benefits when the land slope exceeds 0.2%. As a consequence, controlled drainage has for the most part been limited to very flat land. This is a result of equal emphasis being placed on crop production and water quality goals.

However, water quality benefits can be obtained without being able to control the water table near the soil surface throughout the entire agricultural field. Slowing the water movement from

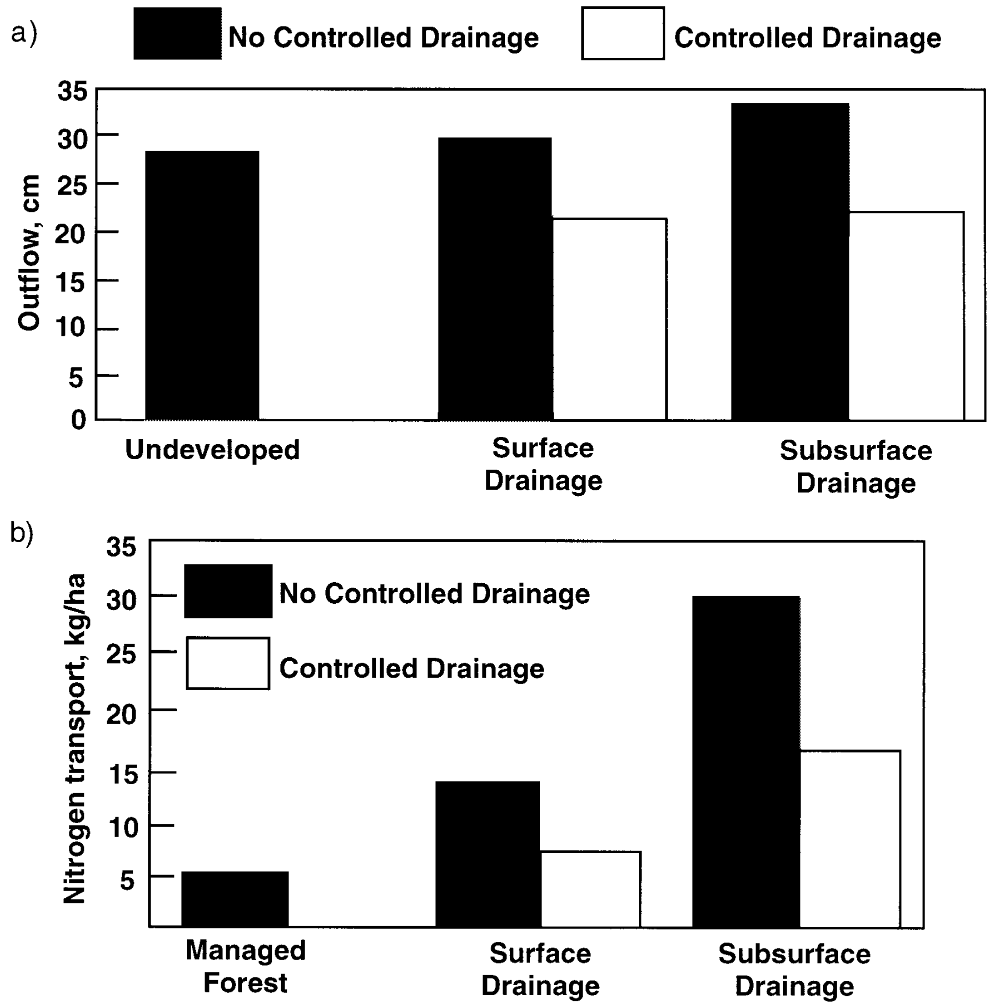


FIGURE 3.17. Results of controlled drainage study represent approximately 125 site-years of data from 14 sites in eastern North Carolina showing: (a) average annual outflows, and (b) average annual nitrogen transport (TKN + NO_3-N) in drainage outflow as measured at the field edge for 14 soils and sites. (*From Evans et al. 1991.*)

the field and maintaining it within 1 m of surface will promote denitrification and reduce nitrate losses to surface water. If management emphasis shifts so that reduction of nitrogen losses to surface water is sufficiently important that adopted practices need not provide a positive financial return to the landowner, then controlled drainage may become more attractive to the landowner than some other control measures.

This is happening in the Neuse River watershed in North Carolina, where farmers are preferring to use controlled drainage wherever possible, as opposed to putting riparian buffers on their ditches. A scientific group in North Carolina (Gilliam et al. 1997a) recommended that average water table elevation throughout the length of the ditch be no lower than 90 cm (36 inches) below the land surface to get credit for using controlled drainage to reduce nitrate losses. However, it is recognized that during some seasons, particularly during planting and harvesting, control may have to be lowered to facilitate agricultural activities.

3.2.4.2 Controlled Drainage in the Midwest

Controlled drainage is not practiced frequently in the Corn Belt of the Midwest, and research on this practice has been limited. There appear to be three reasons why: the undulating, pothole topography, the cool temperatures that limit denitrification when subsurface drainage is greatest, and no sense of urgency to try the technique. The variable relief limits the area where controlled drainage can practically be used because installation of controlled drainage facilities in rolling land is expensive and unnecessary in the view of farmers. Raising the water table in cooler climates to increase denitrification in the late fall and spring when soils are cold and wet would not lead to as much denitrification as has been measured in the southern United States, where most controlled drainage research has been conducted. However the method has been applied with success in Canada, where temperatures are cool, and there has been political pressure to reduce nitrates below 10 mg-N/L (Drury et al. 1996). It seems likely that controlled drainage is a practical alternative in some areas of the Midwest to reduce nitrate discharges to surface waters.

There has been some research on this practice in both Iowa and Ohio. Personal communication with scientists in both of these locations (James Baker in Iowa; Norman Fausey in Ohio) confirms that there is potential to use this practice in the Midwest. The results in very flat land should be very similar to those obtained in Canada and North Carolina. Even in some rolling land with subsurface drainage systems in place, controls can most likely be used to reduce nitrate losses to surface water. The selection of nitrogen control practice(s) should generally be made on a site-by-site basis. The decision tree presented in Figure 3.18 can assist best professional judgment in determining whether controlled drainage or riparian buffers should be considered.

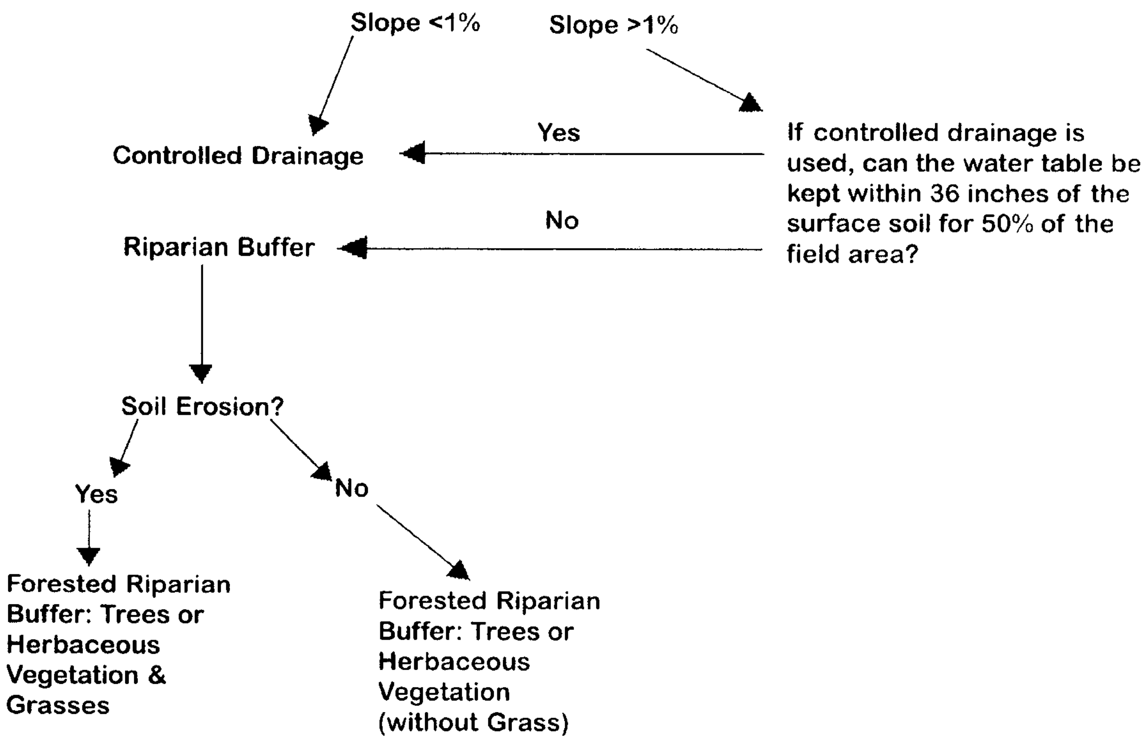


FIGURE 3.18. Decision tree for determining which nitrate control practice to use.

3.3 URBAN NONPOINT-SOURCE CONTROL

Urban areas are well-known nonpoint sources of coastal pollution. Urbanization results in hydrologic changes that increase surface runoff and erosion, and urban areas contain numerous sources of nutrients produced by human activities. A large body of information is available on approaches for reducing nutrients from urban areas. This section provides information on the nature and extent of urban nonpoint-source pollutant control that is possible with current technology, along with scenarios for implementing these controls in the Gulf of Mexico watershed.

The prospects for significantly reducing urban nonpoint-source pollutants, especially N, are not high. However, urban areas do not appear to be major contributors of nonpoint sources of nitrogen to the Gulf. Nitrate concentrations in urban nonpoint sources are generally not high, compared to concentrations in urban point sources or Corn Belt cropland (Tables 1.2 and 3.17). Furthermore, the land area of urban centers compared to rural land in the Mississippi River Basin is small. Several studies have documented that urban areas and lawns have low nitrate concentrations and fluxes relative to agricultural land (Table 3.18), but the comparison depends on the degree of fertilizer use and whether septic systems or central sewer systems are used in the urban setting (Morton et al. 1988; Gold et al. 1990; Petrovic 1990; Miller et al. 1997).

TABLE 3.17. Estimated mean runoff concentrations for land uses, based on the nationwide urban runoff program.

Parameters	Residential	Commercial	Industrial
TKN (mg-N/l)	0.23	1.5	1.6
$NO_3 + NO_2$ (mg-N/l)	1.8	0.8	0.93
Total P (mg/l)	0.62	2.29	0.42
Copper (µg/l)	56	50	32
Zinc (µg/l)	254	418	1,063
Lead (mg/l)	293	203	115
COD (mg/l)	102	84	62
TSS (mg/l)	228	168	108
BOD (mg/l)	13	14	62

Source: USEPA 1993.

TABLE 3.18. Comparison of percolation of nitrate–nitrogen from fertilized and unfertilized urban lawn with fertilized corn cropland.

Treatment Methods	**Flow-Weighted Nitrate-N** *(mg-N/L)*		**Nitrate-N Flux** *(kg-N ha^{-1} yr^{-1})*	
	1997	***1998***	***1997***	***1998***
Agriculture				
Urea-fertilized corn w/ cover crop	15.3	8.1	79.3	41.8
Urea-fertilized corn w/o cover crop	14.9	15.6	73.1	79.8
Manure-fertilized corn	4.2	17.5	20.2	100.0
Home Lawn				
Fertilized	1.6	0.3	9.3	1.9
Unfertilized	0.2	0.2	1.3	1.4
Oak-Pine Forest	0.2	0.2	1.2	1.5

Source: Gold et al. 1990.

The U.S. Environmental Protection Agency (USEPA) has compiled a large amount of information on urban nonpoint sources and on control technologies and their effectiveness. The Nationwide Urban Runoff Program documented the nature and extent of nonpoint-source pollutant problems in urban areas (Table 3.19; USEPA 1977, 1983). The 1990 Coastal Zone Act Reauthorization Amendments (CZARA) required the states to implement management measures to protect coastal waters from urban nonpoint sources of pollution. Management measures are defined as economically achievable measures to control the addition of pollutants to coastal waters, that reflect the greatest degree of pollutant reduction achievable through the application of the best available nonpoint-source pollution control practices, technologies, processes, siting criteria, operating methods, or other alternatives. The USEPA (1993) has specified and evaluated a wide range of management measures for urban nonpoint sources of pollution.

Controlling nonpoint-source pollution from urban areas requires the use of two primary strategies: prevention of pollutant loadings and treatment of unavoidable loadings. The USEPA management measures attempt to address the prevention and treatment of nonpoint-source pollutants from all phases of urbanization and are strongly watershed based. There are approaches for reducing pollutants from existing as well as from new development.

3.3.1 Existing Development

3.3.1.1 STORMWATER RUNOFF

Reducing nonpoint-source pollutants from existing urban areas requires identifying opportunities for local and/or regional pollutant reduction via installation and/or improvements in existing urban runoff control structures. Table 3.19 presents specific management practices for treating runoff from exiting urban developments. Total nitrogen removal for these practices ranges from 5% to 60%. The primary limitations to their effectiveness stem from their inability to remove N from surface runoff. Surface runoff is difficult to treat because it moves rapidly through the landscape, often as channelized flow, reducing the potential for biological attenuation of N.

The total possible reduction of the nonpoint-source load from existing development is limited by the effectiveness of the site-specific practices listed in Table 3.19, multiplied by the percent of total urban runoff that is treated by the practices. For example, an effort in the Anacostia River watershed in Washington D.C., identified 125 sites that would benefit from improvement in storm-water runoff control structures; however, practical considerations limited application of improvements to only 20 of these sites (Schueler et al. 1991). Similarly, in the Loch Raven Reservoir watershed in Baltimore, 24 sites were identified, but it was possible to implement improvements at only 7 of them (Stack and Belt 1989).

A minimal-reduction scenario assumes that runoff is treated at 15% of all possible sites, with a removal efficiency of 30%, resulting in a 4.5% reduction of N loading from urban runoff. A middle-of-the-road scenario assumes that 30% of all possible sites are treated, resulting in a 9% reduction. And a high-reduction scenario assumes that 60% of all possible sites are treated, resulting in an 18% reduction of N loading from urban runoff.

TABLE 3.19. Summary of existing development management practices for controlling sediments (TSS), phosphorus (TP), and nitrogen (TN) in urban runoff.

Management Practices	% Removal *TSS*	*TP*	*TN*	Main Removal Efficiency Factors
Water Quality Inlet—Catch Basin				
Average	15	5	5	Maintenance
Reported Range	10–95	5-10	5-10	Sedimentation storage volume
Probable Range	10-25	5-10	5-10	
Number of Values Considered	2	1	1	
Water Quality Inlet—Catch Basins with Sand Filter				
Average	80	NA	35	Sedimentation storage volume
Reported Range	75–85	NA	30–45	Depth of filter media
Probable Range	70-90	–	30-40	
Number of Values Considered	1	0	1	
Water Quality Inlet— Oil/Grid Separator				
Average	15	5	5	Sedimentation storage volume
Reported Range	10-25	5-10	5-10	Outlet configurations
Probable Range	10-25	5-10	5-10	
Number of References	1	1	1	
Dry Pond Modified into Ed Dry Pond				
Average	45	25	35	Storage volume
Reported Range	5–90	1 0–55	20–60	Detention time
Probable Range	70–90	10–60	20–60	Pond shape
Number of Values Considered	6	6	4	
Dry Pond Modified into Wet Pond				
Average	60	45	35	Pool volume
Reported Range	(-30)–91	10–65	5–85	Pond shape
Probable Range	50–90	20–90	10–90	
Number of Values Considered	11	10	7	
Dry Pond or Wet Pond Modified into Ed Wet Pond				
Average	80	65	55	Pool volume
Reported Range	50–100	50–80	55	Pond shape
Probable Range	50–95	50–80	–	Detention time
Number of Values Considered	1	1	1	
Streambank Stabilization				
Average	NA	NA	NA	
Reported Range	NA	NA	NA	
Probable Range	–	–	–	
Number of Values Considered	0	0	0	
Riparian Forest (assumed same as Vegetated Filter Strip)				
Average	70	50	60	Runoff volume
Reported Range	20–80	30–95	40–70	Slope
Probable Range	40–90	30–80	20–60	Soil infiltration rates
Number of Values Considered	6	3	2	Vegetative cover Buffer length
Wetland (assumed same as Storm Water Wetlands)				
Average	65	25	20	Storage volume
Reported Range	(-20)–100	(-120)–100	(-15)–1	Detention time
Probable Range	50–90	(-5)–80	0–40	Pool shape
Number of Values Considered	14	14	6	Wetland's biota Seasonal Variation

Source: USEPA 1993.

3.3.1.2 ON-SITE SEWAGE DISPOSAL SYSTEMS

The USEPA includes on-site sewage disposal (OSD) systems (or septic systems) under urban nonpoint sources. OSD systems have been identified as major sources of nitrogen in several coastal areas (Tyson 1997; Valiela et al. 1997). Although there are several technologies that can reduce nitrogen loads from OSD systems significantly (e.g., by 50%; see Table 3.20), they are considerably more expensive than conventional OSD systems, which inhibits their use in new and existing developments. However, OSD systems have a turnover time of about 30 years, so there is potential to improve technology and nutrient removal performance systematically.

TABLE 3.20. On-site sewage disposal (OSD) system effectiveness and cost summary.

Management Practices	Effectiveness[1]				Costs[2]	
	TSS (%)	***BOD (%)***	***TN (%)***	***TP (%)***	***Capital[3] ($/House)***	***Maintenance ($/Year)***
Conventional Septic System						
Average	72	45	28	57	$4,500	$70
Reported Range	60–70	40–55	10–45	30–80	$2,000–8,000	$50–100
Probable Range	54–83	30–60	0–58	0–95	$2,000–10,000	$25–110
# Values Considered	7	7	13	1.2	8	4
Mound Systems						
Average	NA	NA	44	NA	$8,300	$180
Reported Range	60–70	40–55	10–45	30–80	$7,000–10,000	$100–300
Probable Range	NA	NA	44-44	NA	$6,800–I1,000	$90–310
# Values Considered	0	0	1	0	4	4
Low-Pressure Systems						
Average	NA	NA	NA	NA	$5,100	$150
Reported Range	60–70	40–55	10–45	30–80	$4,000–6,000	$100–200
Probable Range	NA	NA	NA	NA	$2,800–7,400	$150–150
# Values Considered	0	0	0	0	2	1
Anaerobic Upflow Filter						
Average	44	62	59	NA	$5,550	NA
Reported Range	30–60	50–75	40–75	60–80	$3,000–8,000	$150–400
Probable Range	24–89	46–84	20–75	NA	$3,000–8,000	NA
# Values Considered	6	6	6	0	2	0
Intermittent Sand Filter						
Average	92	92	55	80	$5,400	$275
Reported Range	80–95	90–95	50–65	70–90	$4,000–8,000	$250–400
Probable Range	70–99	80–99	40–75	70–90	$2,300–10,000	$100–440
# Values Considered	7	10	7	2	7	5
Recirculating Sand Filter						
Average	90	92	64	80	$3,900	$145
Reported Range	85–95	85–95	60–85	70–90	$5,000–8,000	$250–400
Probable Range	70–98	75–98	1–-94	70–90	$1,850–9,200	$15–410
# Values Considered	12	15	13	2	5	7
Water-Separation System						
Average	60	42	83	30	$8000	$300

Reported Range	55–70	35–55	70–90	30–55	$5,000–11,000	$300–750
Probable Range	36–75	22–55	68–99	14–42	$5,000–11,000	$300–300
# Values Considered	4	3	6	6	1	1

TABLE 3.20, continued.

Management Practices	Effectiveness[1]				Costs[2]	
	TSS (%)	*BOD (%)*	*TN (%)*	*TP (%)*	*Capital[3] ($/House)*	*Maintenance ($/Year)*
Constructed Wetlands						
Average	80	81	90	NA	$710	$25
Reported Range	60–90	70–90	60–90	30–70	$1,000–3,000	$25–100
Probable Range	50–983	65–97	90–90	NA	$50–350	$25–25
# Values Considered		4	2	0	19	1
Eliminating Garbage Disposals						
Average	37	23	5	2.5	NA	NA
Reported Range	35–40	25–30	5–10	2–3	Negligible	Negligible
Probable Range	37–37	28–28	5–5	2–3	NA	NA
# Values Considered	3	2	2	2	NA	NA
Low-Phosphate Detergents						
Average	NA	NA	NA	50	NA	NA
Reported Range	NA	NA	NA	40–50	Negligible	Negligible
Probable Range	NA	NA	NA	50–50	NA	NA
# Values Considered	0	0	0	2	0	0
Holding Tanks						
Average	NA	NA	NA	NA	$3,900	$1,300
Reported Range	95–100	95–100	95–100	95–100	$4,000–6,000	$1,000–200
Probable Range	NA	NA	NA	NA	$1,220–6,570	$100–2,400
# Values Considered	0	0	0	0	8	12

Note: NA = not available.
[1]Effectiveness values reflect total system reduction, including soil absorption fields.
[2]Costs are in 1988 equivalent dollars.
[3]An average household with four occupants was assumed.
Source: USEPA 1993.

A minimal-reduction scenario assumes that no nitrogen-reducing OSD systems are installed because of their high cost. A middle-of-the-road scenario assumes that 10% of the existing OSD systems are replaced with systems that reduce nitrogen loads by 50%. These systems would be installed in particularly sensitive coastal areas, as specified in the USEPA management measures for OSD systems recommended to achieve compliance under the CZARA (USEPA 1993). This scenario would result in a total nitrogen reduction from OSD systems of 5%, but this reduction needs to be highly weighted because it would occur in the immediate coastal zone. A high-reduction scenario assumes that 50% of all OSD systems have nitrogen-reducing technology, for a total reduction of 25%. It is important to note that any OSD system reductions would require a 30-year implementation time, assuming that all systems are replaced on a 30-year cycle.

3.3.2 New Development

Analysis of nonpoint-source pollutant loads from urban areas is greatly complicated by the fact that new development is inevitable. In many coastal areas, urban and suburban development is rapidly replacing agricultural and natural areas. These land-use and/or land-cover changes must be considered in evaluations of nitrogen delivery to the Gulf of Mexico.

The USEPA management measures suggested to achieve compliance with the CZARA are designed to reduce the impacts of new development. They focus strongly on sediments, specifying, for example, either an 80% reduction in sediment loads or a limit of sediment loads to pre-development levels (USEPA 1993). Given that practices that reduce sediment loadings also reduce nitrogen, but much less effectively (Tables 3.19 and 3.20), new development will result in increased nitrogen loadings to the Gulf over time. Though our scenarios will vary in the extent of the increase, some increase is inevitable and must be accounted for.

The impacts of new development can be greatly reduced with watershed- and site-scale planning. Watershed planning efforts need to identify both areas most likely to cause problems if developed and areas that play an important role in absorbing nutrients (Table 3.21). Site planning efforts are used to avoid high pollutant outputs during construction activities.

No data are available on the effectiveness—i.e., percent reductions—of planning efforts on nutrient outputs from new development. Moreover, predicting nutrient loads associated with new development requires detailed scenarios of land-use change, which can be difficult to evaluate. For example, conversion of agricultural land to urban and suburban uses, which is common in the Gulf of Mexico watershed, can result in decreases in nitrogen outputs, especially if state-of-the-art nonpoint-source pollutant controls are used in the new development. Alternatively, conversion of forest and native grassland to urban and suburban uses will increase nitrogen outputs. Calculating reduction scenarios requires: (1) obtaining land-use/land-cover change scenarios for the watershed; (2) computing the change in nitrogen delivery that the scenarios would cause; and (3) computing the effect of instituting state-of-the-art nonpoint-source pollutant control technologies on this change.

3.4 POINT-SOURCE CONTROL

Municipal wastewater is the primary point-source discharge of nitrogen to waterways in the United States. In the Mississippi River Basin, it accounts for an estimated 200,000 metric tons/year of nitrogen discharged to streams and rivers in the basin (Goolsby et al. 1999). There are two basic methodologies for controlling nitrogen and phosphorus from wastewater treatment plants, generally referred to as tertiary treatment: (1) environmental technology and (2) ecotechnology involving wetland treatment systems.

3.4.1 Environmental Technology

Nitrogen, more specifically nitrate–nitrogen, can be removed from the water column by a number of engineered, treatment plant technologies that involve chemical, physical, and biological processes. These technologies rely on the controlled use of chemicals and of mechanical energy within structured environments, such as concrete or earthen containers. Generally, the engineered technologies are labor intensive, requiring continuous monitoring and management of the treatment factors; but on the positive side, they require little land.

Because of the need for fixed containers, engineered technologies are limited to small fluctuations in flow. For example, during storm events when flow rates often double or triple within urban collection systems, the capacity of the receiving treatment plant is often exceeded and wastewater flow is bypassed to avoid damage to the physical plant or disruption of the treatment process. Consequently, engineered technologies are most efficiently applied in situations where flow variations are limited, such as in industrial or domestic sewerage applications. Applying these technologies to urban or agricultural runoff, which is subject to large variations in flow,

TABLE 3.21. Step-by-step guide to watershed management for new urban sources of pollution.

1. **Delineate and map watershed boundary and sub-basins within the watershed.**
2. **Inventory and map natural storm-water conveyance and storage systems.**
3. **Inventory and map man-made storm-water conveyance and storage systems.**
   This includes all ditches, swales, storm sewers, detention ponds, and retention areas and such information as their size, storage capacity, and age.
4. **Inventory and map land use by sub-basin.**
5. **Inventory and map detailed soils by sub-basin.**
6. **Establish a clear understanding of water resources in the watershed.**
   Analyze water quality, sediment, and biological data. Analyze subjective information on problems (such as citizen complaints). Evaluate waterbed use impairment-frequency, timing. seasonality of problem. Conduct water quality assessment-low flow, seasonality.
7. **Inventory point and nonpoint sources of pollution in the watershed.**
   This includes pollutant identification, location, loadings, flow, capacity, and:
   - land-use/loading-rate analysis for storm water;
   - sanitary survey for septic tanks; and
   - dry-flow monitoring to locate illicit discharges.
8. **Identify and map future land use by sub-basin.**
   Conduct land-use loading-rate analyses to assess the potential effects of various land-use scenarios.
9. **Identify planned infrastructure improvements—5 years, 20 years.**
   Coordinate and schedule storm-water management deficiencies with other infrastructure or development projects.
10. **Analysis.**
    Determine the infrastructure and natural resource management needs within each watershed.
11. **Set resource management goals and objectives**
    Before corrective actions can be taken, a resource management target must be set. The target may be defined in terms of water quality standards (e.g., attainment and preservation of beneficial uses) or other local resource management objectives.
12. **Determine pollutant reduction (for existing and future land uses) needed to achieve water quality goals.**
13. **Select appropriate management practices for point and nonpoint-sources that can be used to achieve the goal.**
    Evaluate the effectiveness of pollutant removal, landowner acceptance, financial incentives and costs, availability of land, operation and maintenance needs, feasibility, and availability of technical assistance.
14. **Develop a watershed management plan.**
    Since each watershed's problems are unique, each management plan will be different. However, all watershed plans should include such common elements as:
    - an existing and future land-use plan;
    - a master storm-water management plan that addresses existing and future needs;
    - a wastewater management plan, including septic tank maintenance programs; and
    - an infrastructure and capital improvements plan.

Source: Livingston and McCarron 1992, as cited in USEPA 1993.

dramatically increases the capital cost because of the need for large containers, which are infrequently used. Further, treating storm-water runoff with engineered technologies is complicated by the means of capture. For example, agricultural tiles are widely spaced, and the distance between tile systems is extensive. Collecting and treating the effluent from these systems would be feasible, but would require expensive intercepting sewers or numerous small treatment plants.

Engineered technologies have proved to be successful in treating wastewater, with the exception of sewer systems that combine both wastewater and storm water. In the latter case, large, expensive reservoirs have had to be constructed to modulate the flow sufficiently to accommodate the technology. For industrial wastewater, the engineered application is most suitable, assuming the process is well defined and the waste nitrogen load is uniformly distributed. Engineered technologies also could work in agricultural fields, particularly with regard to contained feedlot operations, for example, for cattle, swine, or chickens. Less appropriate are those applications where nitrate concentrations are low, where storm-water runoff is mixed with wastewater, and where flow streams are dispersed over a wide area.

3.4.1.1 Physical/Chemical Processes

The three commonly used physical/chemical processes for controlling nitrogen in wastewater are air stripping, breakpoint chlorination, and ion exchange. These processes are relatively expensive and require careful monitoring and control (Metcalf and Eddy Inc. 1991). Air stripping requires the adjustment of pH through the addition of lime, causing ammonia gas to be formed, which is then released by mechanical stirring or aeration. In the breakpoint chlorination process, chlorine is added to the water, which strips the hydrogen atoms off the ammonia molecule, releasing nitrogen gas. A significant drawback to this process is that highly toxic residual chlorine often finds its way into the receiving stream. In the ion exchange process, ions of one species are displaced from an insoluble ion of a different species; a resin is stirred into the water, which attracts the ammonium cations; the resin then precipitates and can be removed and regenerated.

3.4.1.2 Biological Processes

The activated sludge process is the principal biological mechanism for removing contaminants, including nitrogen from municipal and industrial sources. This process relies on using microbes to consume the unwanted substances, flocculate, settle, and form sludge, which is removed from the bottom of the container. Typically, the activated sludge process results in the conversion of ammonium–nitrogen to nitrate, through nitrification. Nitrate is then released to the receiving surface water. However, the process can be taken further to remove nitrate through denitrification. These processes are reasonably stable, require little land, and are generally cheaper than the physical/chemical methods.

The activated sludge process may contain two stages—the first resulting in the conversion of ammonium to nitrate, and the second in the conversion of nitrate to nitrogen gas. The process can be carried out (1) in a suspended-growth medium, which requires the suspension of microbes in the wastewater by means of mechanical stirring or aeration; or (2) on an attached-growth medium, which involves hard substrates, such as gravels or synthetic substrates, on which the microbes attach and grow. The removal rate of nitrate is in the range of 80–90%.

A variety of innovative variations to the basic activated sludge processes have been tested and explored in recent years. For instance:

- Simultaneous nitrification and denitrification can be accomplished in a single bioreactor by using both partly and completely submerged biofilms (Watanabe et al. 1994).

- In moving-bed biofilm reactors, the biofilm grows on small plastic elements shaped like short pipes with a cross inside (Rusten et al. 1995).
- A recently designed and tested biological nutrient-removal divides aeration basins into three functional zones—anaerobic, anoxic, and aerobic—facilitating the removal of nitrogen from the flow stream process (DeBarbadillo et al. 1995).
- Integrated fixed-film media have been used to increase denitrification in anaerobic sections of anaerobic treatment plants (Randall and Sen 1996).
- A biofilm-sequencing batch reactor uses four phases to remove nitrogen: anaerobic, aerobic, anoxic, and aerobic. Denitrification accomplished in the anoxic phase has reduced nitrogen by 87% (Garzón–Zuniga and González 1996).
- Single-basin lagoon systems are achieving nitrification and denitrification by turning the aerators on and off at set intervals. This method yielded a nitrogen removal rate of 81% and a 95% ammonium–nitrogen conversion (Rothberg et al. 1993).

3.4.1.3 FEASIBILITY

Using engineered processes to remove nitrate–nitrogen from point sources is feasible where land is limited (such as in urban areas), when nitrate is concentrated in the flow stream, and when the flow stream is reasonably uniform with respect to time. Although these conditions are met for some municipal, industrial, and confined agricultural applications, the percentage of nitrate–nitrogen coming from treatment plants in the Mississippi River Basin is a small portion of the total load to the Gulf of Mexico. Consequently, implementing denitrification at existing plants would be ineffective and costly.

Applying these technologies to urban and agricultural nonpoint-source runoff is equally infeasible because the flow paths are widely distributed, reasonably dilute, and highly variable. To meet peak demands, enormous capital investment would be required to provide for the necessary tankage or large reservoirs to moderate flow fluctuations. Further, production of the needed chemicals and electrical energy would degrade the environment by emitting greenhouse gases and solid waste by-products. Far more efficient and environmentally safe technologies can be found through ecotechnology.

3.4.2 Ecotechnology—Treatment Wetlands

Countless studies have demonstrated the capacity of created and natural marshes to be sinks for nitrogen and other contaminants in wastewater (e.g., Fetter et al. 1978 in Wisconsin; Kadlec and Tilton 1979 and Kadlec 1983 in Michigan; Dierberg and Brezonik 1984, 1985, and Dolan et al. 1981 in Florida; Knight et al. 1987 in North Carolina; and Spieles and Mitsch (2000) in Ohio). There are now hundreds of documented wastewater wetlands in the United States and Europe. Results of many of these constructed wetlands for nutrient retention have been summarized in a data base maintained by the USEPA. Summaries of those data are given by Kadlec and Knight (1996). This section presents some design considerations for these treatment wetlands and contrasts them with natural and nonpoint-source wetlands described above.

3.4.2.1 SURFACE OR SUBSURFACE FLOW

Generally, constructed treatment wetlands are designed for either surface flow over the substrate or subsurface flow through a substrate (Figure 3.19). Surface-flow wetlands, though generally less effective in removing some pollutants at first, are closer in design to natural wetlands and are less prone to clogging and, therefore, require less maintenance. Subsurface flow through artificial wetlands can be through soil media (*root-zone method*) or through rocks or sand (*rock-reed filters*), with the flow in both cases 15–30 cm below the surface (Wieder et al. 1989). In a survey of several hundred wetlands built in Europe for sewage treatment in rural settings, Cooper and Hobson (1989) report that gravel is used in combination with soil, but that the substrate remains the greatest uncertainty in artificial-reed (*Phragmites*) wetlands used for water quality enhancement. Constructed wetlands with subsurface flow have the advantage of requiring a smaller area for the same retention of chemicals, but they are prone to clogging if overloaded.

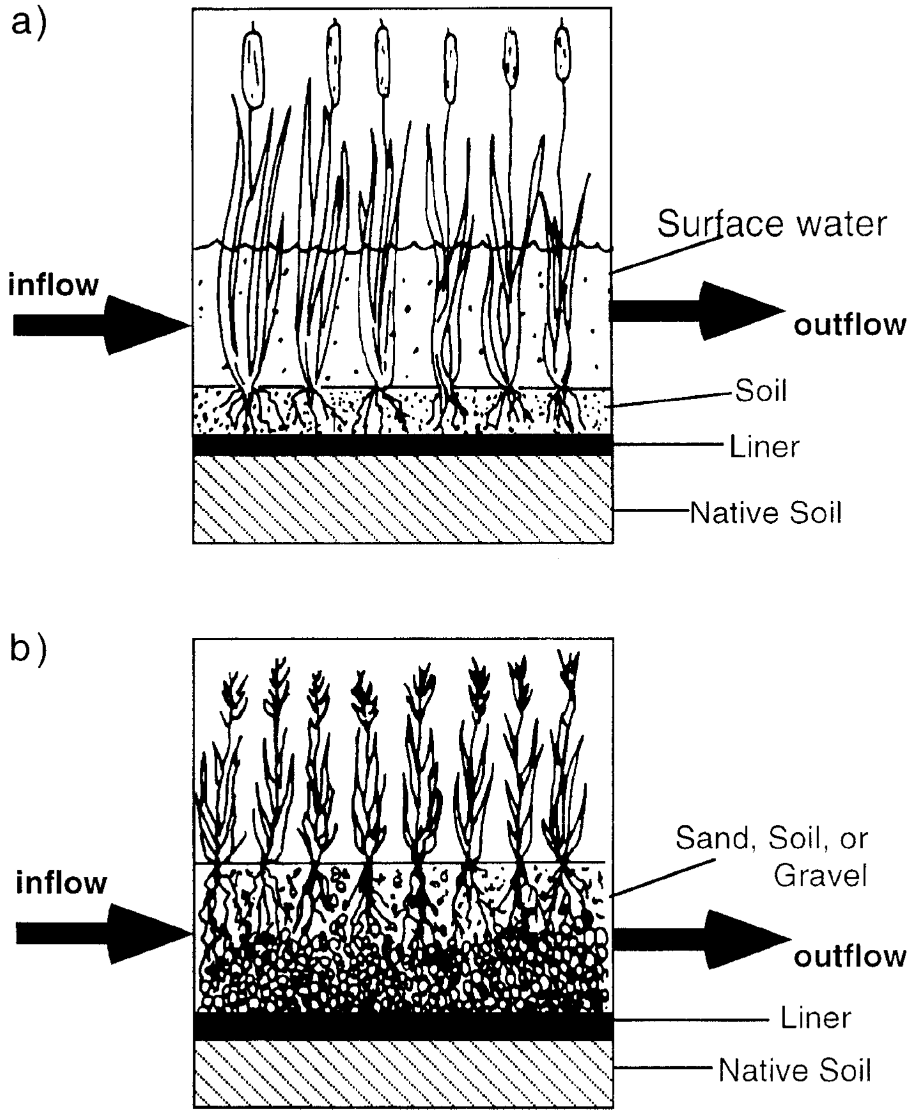


FIGURE 3.19. Schematic showing differences between (a) surface-flow and (b) subsurface-flow treatment wetlands. (*From Knight 1990.*)

3.4.2.2 TREATMENT WETLAND DESIGN

The nutrient retention capacity of wastewater wetlands has been well documented (reviewed by Kadlec and Knight 1996). A hypothetical nitrogen mass balance for a moderately loaded wastewater wetlands, as illustrated in Figure 3.20, shows that wastewater wetlands are capable of routinely removing 100–300 g-N m^{-2} yr^{-1}, a rate much higher than wetlands used for nonpoint-source control. Kadlec and Knight (1996) point out that the role of vegetation uptake in the nitrogen budget is not trivial and can be 25% or more of the retention. However, only a fraction of that nitrogen is permanently buried in the sediments. In addition, the rates of nitrification and denitrification greatly exceed the rates that would be estimated from only a water quality inflow–outflow analysis. Kadlec and Knight (1996) estimated that the true rate of denitrification in wastewater wetlands, based on rate constants rather than water quality analyses, is on the order of 280 g-N m^{-2} yr^{-1} (Figure 3.20), a rate far in excess of those estimated for most natural wetlands and riparian forests.

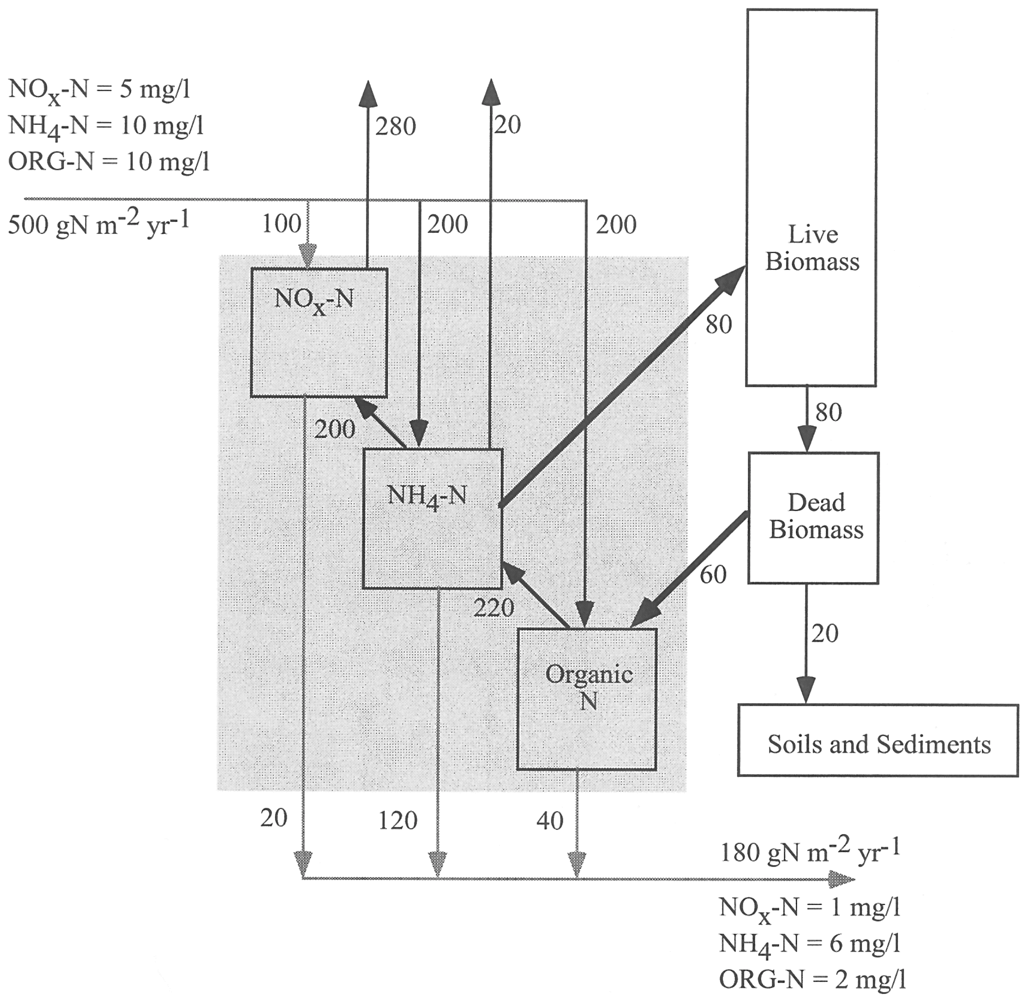


FIGURE 3.20. Hypothetical nitrogen fluxes in wastewater treatment wetlands based on hydrologic loading of 5.5 cm/day and first-order decay rates determined from multiple sites. (*From Kadlec and Knight 1996*.)

Table 3.22 summarizes nitrogen removal efficiency of wastewater wetlands from the North American data base. Removal efficiencies range from 46% removal for constructed surface-flow treatment wetlands to 72% for natural surface-flow wetlands. Removal rates averaged from 69 to 570 g-N m^{-2} yr^{-1}, with the low number for natural wetlands receiving wastewater, and the high number for constructed subsurface wastewater wetlands. These numbers are far in excess of what occurs in natural wetlands, where nitrogen retention rates are generally in the range of 0–40 g-N m^{-2} yr^{-1}. Because it is very unlikely that natural wetlands can continue to be used to treat wastewater because of wetland protection laws, it can be assumed that constructed wastewater wetlands are the primary wetland type for controlling nitrates from point sources. With loading on the order of 300–900 g-N m^{-2} yr^{-1}, nitrogen flows could be expected to be reduced by about 50% with some consistency. The high rates of nitrogen removal that are possible with constructed wetlands treating domestic wastewater suggest that these systems are efficient alternatives for controlling nitrogen from point sources. The generally lower costs of these wastewater treatment wetlands, as alternatives to the more costly environmental technology described above, add to their desirability as nitrogen control systems.

TABLE 3.22. Nitrate and total nitrogen removal rates and efficiency of natural and constructed wastewater wetlands as averaged from a number of systems in North America.

Parameters	Natural Wetlands	Constructed Wetlands ***Surface Water***	 ***Subsurface Water***
Nitrate + Nitrite N			
Inflow Concentrations *(mg/L)*	4.30	1.90	109
Outflow Concentrations*(mg/L)*	0.29	1.21	94.5
Loading Rates (g m^{-2} yr^{-1})	52	29	5,767
Removal Rates (g m^{-2} yr^{-1})	40	13	547
Efficiency (%)	77.5	44.4	9.4
Total N			
Inflow Concentrations *(mg/L)*	10.2	8.08	41.4
Outflow Concentrations *(mg/L)*	2.3	4.58	12.1
Loading Rates (g m^{-2} yr^{-1})	96	277	1,058
Removal Rates (g m^{-2} yr^{-1})	69	126	569
Efficiency (%)	71.9	45.6	53.8

Source: Kadlec and Knight 1996.

3.5 CONTROL OF ATMOSPHERIC NO_X

3.5.1 Stationary Sources

Control technologies in stationary sources reduce NO_x ($NO + NO_2$) gaseous emissions during either combustion or post-combustion processes (Figure 3.21). The first type of technology modifies the combustion process, such as low-NO_x burners and gas reburning. The second type involves post-combustion removal of NO_x generated during the combustion process, including selective noncatalytic NO_x reduction and selective catalytic reduction. Most of these technologies can be used on new boilers or retrofitted to existing boilers.

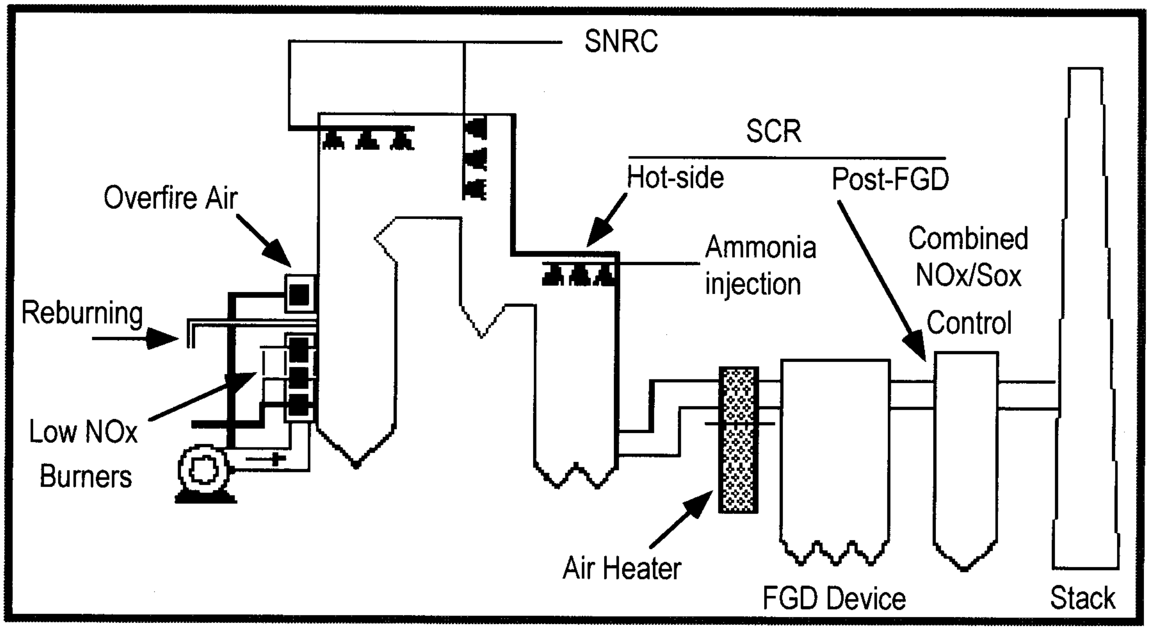


FIGURE 3.21. Combustion and post-combustion NO_x control options for stationary sources. NOTE: SNCR = selective non-catalytic reduction; SCR = selective catalytic reduction; FGD = flue-gas desulfurization. (*From Tavoulareas and Charpentier 1995*.)

3.5.1.1 MODIFIED COMBUSTION PROCESSES

Low-NO_x burners (LNBs) stage the combustion process to control the coal–air mixture at each stage (combustion zone). An oxygen-deficient region and delayed combustion of coal are created by introducing auxiliary air to the outside of the firing zone, thus limiting the availability of oxygen to react with the nitrogen in the coal. Approximately 30–55% of the NO_x can be reduced by this method (Tavoulareas and Charpentier 1995).

Gas reburning involves introduction of up to 20% of total fuel input from natural gas above the main combustion zone. When flue gas containing NO_x drifts upward from the main combustion zone, NO_x is reburned with gas in this upper level, fuel-rich zone and converted into molecular nitrogen. LNBs plus reburning can reduce NO_x by up to 72% of NO_x (USEPA 1996b).

3.5.1.2 POST-COMBUSTION PROCESSES

Selective noncatalytic reduction (SNCR) is a post-combustion process that injects a nitrogen-based reducing agent (such as urea and ammonia) into the flue gas to reduce NO_x to molecular nitrogen and water. The general reaction is as follows:

$$NO_x + NH_3 + O_2 + H_2O + (H_2) \rightarrow N_2 + H_2O + \text{Heat}$$

The reagent is usually injected at the top and backpass of the boiler, where internal temperature is between the optimal reaction range (870–1200°C). SNCR can reduce NO_x by up to 35–50% of NO_x.

Selective catalytic reduction (SCR) is similar to SNCR in that it injects ammonia in the flue gas, but at a lower temperature (340–380°C) with a catalyst (a vanadium/titanium formulation and zeolite materials). The NO_x in the flue gas reacts with NH_3 that is adsorbed onto the active sites on the catalysts to form molecular nitrogen and water. The SCR technology was developed in the United States, but has been more aggressively implemented overseas (USEPA 1997a). It may provide the greatest opportunity for

NO_x reduction, as it is capable of decreasing NO_x emissions by more than 90% (STAPPA & ALAPCO 1994).

3.5.2 Mobile Sources

Automobile NO_x emission control technology has made significant advances in the past several years. Many of the technologies discussed below can reduce NO_x emissions beyond the level of control required by Tier 1 standards, and have been already used on current Tier 1, national low-emission vehicles (LEVs) and California LEV technology vehicles (USEPA 1998a). These technologies include improvements in base engine-out emissions, air–fuel ratio control, fuel delivery and atomization, and exhaust after-treatment (Table 3.23).

3.5.2.1 BASE ENGINE IMPROVEMENT

Engine-out NO_x emissions result from high combustion temperatures. Exhaust gas recirculation (EGR), multiple valves with variable valve timing and use of "fast burn" combustion chamber designs provide improved thermal efficiency and lower NO_x emissions. A leak-free exhaust system can also be used to prevent outside ambient air from being drawn into the system that otherwise would increase emission level. These technologies provides 3-10% reductions of NO_x from current Tier 1 standards (Table 3.23). EGR can result 15-20% reduction in engine-out emissions.

TABLE 3.23. Feasible technologies for NO_x emission reductions (from Tier 1 levels) for mobile sources.

Technology	% NO_x Reduction
Modifications to combustion chamber	3–10
Multiple valves with variable valve timing	3–10
Increased EGR (including electronic control)	$\geq$ 10
Improved A/F control (i.e., improved HEGO, improved power-train control module microprocessor, faster fuel injectors, transient adaptive fuel control algorithms, dual HEGO, and improved calibration)	20
Universal exhaust gas oxygen (UEGO) sensor	23–35
Air/fuel control in individual cylinders	3
Catalyst improvements (thermal stability, washcoat, cell densities)	10
Increased catalyst loading and volume	20
Advanced catalyst designs (tri-metal, multi-layered)	30–57
Close-coupled catalysts	0–10
Electrically-heated catalysts	5–10

NOTE: In general, these percentages cannot be simply summed to achieve a total emission reduction when more than one emission control technology is being applied.
Source: USEPA 1998a.

3.5.2.2 IMPROVEMENTS IN AIR-FUEL RATIO CONTROL

These technologies use dual-heated exhaust gas oxygen (HEGO) sensors, universal exhaust gas oxygen (UEGO) sensors, individual cylinder air/fuel (A/F) control, adaptive fuel control systems, or electronic throttle control systems to aid in controlling air–fuel mixing for complete combustion (as close to stoichiometric operation as possible), thus maximizing the efficiency of the three-way catalyst to reduce NO_x. These improvements can reduce NO_x emissions by 35%.

3.5.2.3 IMPROVEMENTS TO EXHAUST AFTER-TREATMENT SYSTEMS

Many important advances have been made over the last five years in exhaust after-treatment systems. Improvements to the catalyst thermal stability, washcoat, cell density, and multi-layered designs can result in up to a 57% NO_x reduction from the Tier 1 level.

3.5.2.4 ADVANCED TECHNOLOGIES

In addition to the above improvements that have been already found in conventional vehicles, advanced technologies providing even better emission control are being developed on ultra-low-emission vehicles (ULEVs) and zero-emission vehicles (ZEVs). These developments include vehicles powered by compressed natural gas (CNG), battery, hybrid propulsion system (gasoline powered engine plus electric motor), and fuel cells promising a very low or zero emission.

3.5.3 Regulatory Issues

NO_x emission is mainly regulated by the Clean Air Act Amendments (CAAA) of 1990. Title I of the CAAA requires states to regulate in ozone nonattainment areas and ozone transport regions: (1) existing major stationary sources of NO_x to apply reasonably available control technology; and (2) new or modified major stationary sources of NO_x to offset their new emissions and to install controls representing the lowest achievable rate.

Title II of the CAAA calls for reductions in motor vehicle emissions. It sets specific emission standards, known as Tier 1, for light-duty vehicles (LDVs) and light-duty trucks (LDTs) made during and after 1994. Title II further requires the USEPA to study whether more stringent emission standards, known as Tier 2, should be required beginning with the 2004 model year.

Title IV of CAAA was designed to reduce harmful effects of acid deposition by limiting the allowable emissions of sulfur dioxide and nitrogen oxide. It requires NO_x emission reduction from coal-fired utility sources through a two-phased program. Phase I reduces annual NO_x emissions by more than 0.4 million tons/year between 1996 and 1999, while Phase II sets further limitations for various boilers beginning in the year 2000.

3.5.3.1 RECENT DEVELOPMENTS

Over the past two years, the USEPA has released and proposed several rules and regulations aimed to reduce NO_x and other atmospheric emissions under the CAAA. In December 1996, the USEPA released a new rule on Phase II of Title IV of the CAAA that sets lower emission limits for Group 1 burners, and established limitations for Group 2 boilers based on NO_x control technologies that are comparable in cost to LNBs applied in Phase I (Table 3.24). The new rule will reduce about 1.17 million tons of NO_x emissions per year beginning in the year 2000 at a 20% removal efficiency and an average cost-effectiveness of $229 per ton of NO_x removed. This represents a 15% percent reduction of NO_x emissions from utilities, or 5% of total NO_x from all sources. The emission limitation set by this rule is very cost-effective compared to other source controls such as those for industrial and automobile sources (Figure 3.22).

TABLE 3.24. Coal-fired boiler types and the best continuous control systems used by EPA to establish NO_x emission limits under Title IV of the Clean Air Act Amendments.

Coal-Fired Boiler Types	Characteristics	Control Systems
Group 1		
Dry-bottom wall-fired boilers	Burners are located along the furnace wall.	Low NO_x burners
Tangentially fired boilers	Burners are located in the furnace corners.	
Group 2		
Cell burner-fired boilers	Dry-bottom boilers with arrays of circular burners forming a cell mounted on the furnace wall.	Plug-in and non-plug-in
Wet-bottom boilers	Ash is converted into molten slag at bottom of furnace by high internal temperature.	Gas-reburning SCR
Cyclones boilers	Wet-bottom boilers that burn fuel in horizontal water-cooled cylinders.	Gas reburning SCR
Vertical-fired boilers	Vertically oriented circular burners.	Combustion controls
Stoker boilers and fluidizedbed combustion	Low NO_x emission by design.	High cost

Source: USEPA 1996a.

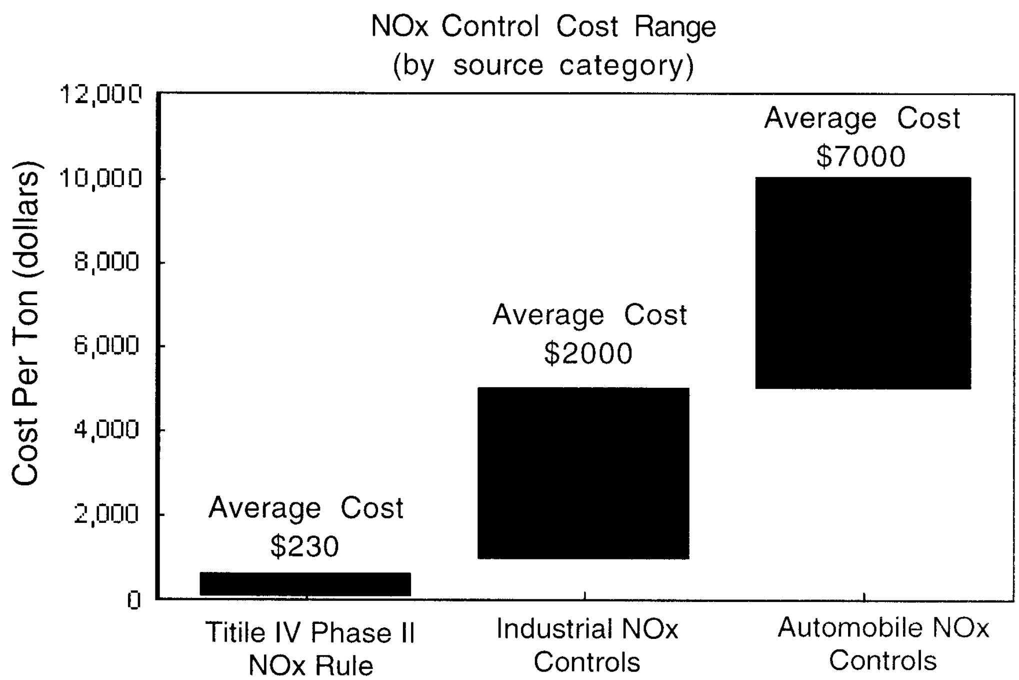


FIGURE 3.22. Stationary and mobile NO_x control cost range by source category. (*USEPA 1996a.*)

On Sept. 24, 1998, the USEPA announced a new anti-smog plan that calls for reducing NO_x emissions by 1.1 million tons annually, or 28% percent overall, in 22 states and the District of Columbia by 2007 (USEPA 1998b). The controls must be in place by 2003. Ohio, the leading state in NO_x emissions from utility power plants, will need to reduce NO_x emissions by 36% by 2003, and 85% by 2007.

According to the Tier 2 report sent to Congress (USEPA 1998a), the USEPA will most likely propose new emission standards at the end of 1999. However, the proposed Tier 2 standards will not be effective before the 2004 model year. Under current vehicle emission control technologies, including tighter air–fuel control and better catalyst designs, the USEPA estimates that NO_x emissions can be reduced by 80% relative to current Tier 1 vehicles (Table 3.25). Meanwhile, under the national low-emission vehicle (LEV) program, a voluntary agreement among automakers and northeastern states will ensure vehicles meet cleaner LEV standards by model year 2001. A LEV-standard vehicle will reduce NO_x by 50% relative to Tier 1 vehicles.

TABLE 3.25. List of potential Tier 2 technologies and associated emission reductions of NO_x for mobile sources.

Technology	% Emission Reduction
Improved A/F Control	20
Increased Catalyst Volume and Loading	20
Improved Catalyst Washcoat/Substrate	10
Close-Coupled Catalyst	10
Advanced Catalyst Design	50
Increased EGR	20
Total[1]	80

[1]Total NO_x reduction = 100% – (100%–20%)*(100%–20%)*(100%–10%)*(100%–10%)*(100–-50%)*(100%–20%) = 80%.
Source: USEPA 1998a.

New emission standards were also set for heavy-duty diesel engines (HDEs) used in trucks and buses that represent about one-quarter of the mobile source NO_x emissions. The new rule requires all HDEs made after 2004 to have a 50% lower NO_x level relative to 1998–2003 model year NO_x standards (USEPA 1997b). This will result in a reduction of 1.1 million tons/year by the year 2020.

3.6 MISSISSIPPI DELTA DIVERSIONS

The hypoxia in the offshore zone of the Gulf of Mexico is likely to be at least partly due to the separation of the Mississippi River from its deltaic plain. Whereas the river once spread out over the delta during flood periods, it is now mostly shunted directly to the sea. In historic times, a considerable amount of water flowed out of the main channel. As one calculation supporting this contention, Kesel et al. (1992) constructed a sediment budget for the lower Mississippi River for the period 1880–1911. They reported that below the Red River, about 26% of the sediment was retained by the delta.

There has been controversy as to the efficacy of diverting river water back into coastal wetlands for nutrient retention (Turner and Rabalais 1991; Rabalais et al. 1994; Day et al. 1997, 1999; Turner 1998). Yet wetlands and shallow-water bottoms with anaerobic sediments are natural sinks for nutrients (Hatton et al. 1982; Sharp et al. 1982; Reddy et al. 1993; see also previous sections), and represent a viable mechanism for decreasing the nutrient load of river water prior to reaching offshore. If river water could be reintroduced to the backwaters, coastal wetlands and shallow inshore bodies of water before its discharge to the Gulf of Mexico, a natural "downstream" pollution control system could be created, augmenting efforts to reduce nutrient inputs from the upstream Mississippi River Basin.

Several case studies are presented in this section to demonstrate the nutrient and sediment dynamics that occur in Louisiana estuaries in response to the addition of Mississippi River water. Among the effects that might occur as a result of this diversion, we hypothesize the following:

1. suspended sediments, nitrate, and total inorganic nitrogen will rapidly assimilate in diverted water;
2. ammonium and organic nitrogen will increase;
3. there will be relatively lower uptake of phosphorus and silicon, resulting in a decrease in the N:P and N:Si ratios in waters going to the Gulf; and
4. diversions will lead to the creation of new wetlands and greater maintenance of existing wetlands.

3.6.1 The Mississippi Delta

The Mississippi River delta formed over the past 6,000–7,000 years as a series of overlapping delta lobes. The coast has often been described in terms of a series of hydrologic basins that are separated largely by current or abandoned distributary channels. The larger delta is made up of two physiographic units: the active deltaic plain to the east and the Chenier plain to the west. Active deltaic lobe formation took place in the deltaic plain. The Chenier plain is a series of old beach ridges formed by westward downdrift of sediments.

The coast is characterized by a series of vegetation zones (saline, brackish, intermediate, and fresh marshes and forested wetlands, from the coast inland) that run roughly parallel to the coast and are determined primarily by salinity. Changes in these zones over the past half century have been described in a series of four vegetation maps (O'Neil 1949; Chabreck 1972; Chabreck and Lins-combe 1982, 1988).

From the 1930s until the present, there has been a dramatic loss of wetlands in the Mississippi Delta, with estimates as high as 100 km^2 per year (Gagliano et al. 1981). Land loss rates were highest in the 1960s and the 1970s and have declined since, although rates remain high (Britsch and Dunbar 1993). An understanding of the causes of this land loss is important not only for developing effective management plans to deal with land loss but also for understanding the relationships among land loss, water quality, and offshore hypoxia.

A number of factors have been linked to land loss, including elimination of riverine input to most of the coastal zone due to construction of flood-control levees along the Mississippi River, altered wetland hydrology mostly due to canal construction, saltwater intrusion, wave erosion along exposed shorelines, high subsidence rates, and sea level rise (see Boesch et al. 1994 for a review of these issues). Most studies have concluded that land loss is a complex interaction of these factors, acting at different spatial and temporal scales (e.g., Day and Templet 1989; Boesch et al. 1994; Day et al. 1995, 1997).

3.6.2 Nitrogen Dynamics in Deltaic Wetlands and Shallow Coastal Waters

Various studies have reported rapid reduction of nitrite plus nitrate ($NO_2 + NO_3$) in estuarine environments, with much of it due to denitrification and other processes (Koike and Hattori 1978; Khalid and Patrick 1988; Lindau and DeLaune 1991; Nowicki et al. 1997). Jenkins and Kemp (1984) reported that up to 50% of $NO_2 + NO_3$ introduced into the Patuxent River estuary underwent denitrification. Vascular plants as well as algae incorporate $NO_2 + NO_3$ into cellular mass. Nitrate reduction to ammonium has also been found to occur (Smith et al. 1982). Sorenson (1978) found as much as 50% of nitrate applied to marine sediments can be reduced to ammonium. These processes are biologically driven and, therefore, are positively correlated with temperature. Denitrification takes place when anaerobic sediments are present. This is the case for coastal wetlands and practically all shallow inshore waters of the Louisiana coastal zone.

Most estuaries are sources for ammonium due to its regeneration during the decomposition of organic matter (Kemp and Boynton 1984), as well as reduction of nitrite and nitrate to ammonium (Sorenson 1978). Numerous studies have shown the net mobilization of ammonium–nitrogen by benthic sediments (Koike and Hattori 1978; Blackburn 1979; Callender and Hammond 1982; Teague et al. 1988). The relatively shallow water depths and rapid settling rates and bacterial utilization result in fairly short residence times for organic material in estuarine waters (Moran and Hodson 1989). Even though ammonium increases, TIN (total inorganic nitrogen) decreases due to denitrification are much higher than regeneration.

Another permanent loss of nitrogen is through burial. Relative sea level rise in coastal Louisiana is approximately 1 cm yr^{-1} (Penland and Ramsey 1990), which is partly compensated for by an accretion rate of 0.7–0.9 cm yr^{-1} (Cahoon and Turner 1989; DeLaune et al. 1989). DeLaune et al. (1981) reported that nitrogen burial in wetlands accreting at a rate of 0.75 cm yr^{-1} was 13.4 g-N m^{-2} yr^{-1}.

3.6.3 Case Studies—Mississippi River Diversions

Three case studies in Louisiana where Mississippi River water has been diverted through shallow water bodies and over wetlands illustrate the effects of diversion of nutrients.

3.6.3.1 The Caernarvon Freshwater Diversion

The Caernarvon diversion is the largest of six diversions currently in operation on the Lower Mississippi River below New Orleans. The structure is located on the east bank of the Mississippi, and water is diverted through several gates so that it can be passed at low river stages (Figure 3.23). The water-control structure has the capability of passing 226 m^3 sec^{-1} of water. Freshwater discharge began in August 1991 and has ranged between 21 and 212 m^3 sec^{-1} (Lane and Day 1999). The diversion delivers water into Breton Sound estuary, which consists of 1,100 km^2 of fresh, brackish, and saline wetlands and many small, shallow water bodies. Breton Sound wetlands were initially formed several thousand years ago as part of the Plaquemines–St. Bernard Delta complex (Scruton 1960). Since then, approximately half of the original wetlands have submerged due to subsidence, as described by Penland et al. (1988).

Several natural crevasses have been documented in the region surrounding Caernarvon, which delivered large amounts of river water into Breton Sound estuary, before the Mississippi River levee system was constructed (Russell 1936; Davis 1993). During the great flood of 1927, the Mississippi River levee at Caernarvon was blown up to relieve New Orleans from possible flooding (Barry 1997). The resulting crevasse was 979 m wide and diverted up to 9,200 m^3 sec^{-1} of water, equal to half of the mean flow of the Mississippi River for four months (Davis 1993; Barry 1997). During the 1927 flood, a layer of river sediments up to 40 cm thick was deposited over an area of approximately 10 km by 15 km. The historical record, therefore, indicates that the Breton Sound estuary has experienced massive periodic inputs of Mississippi River water as part of its evolution to its current ecological state. Villarrubia (1998) reported 164 ha of new marsh has formed in Breton Sound estuary since 1991, and existing wetlands have high rates of accretion on the marsh surface.

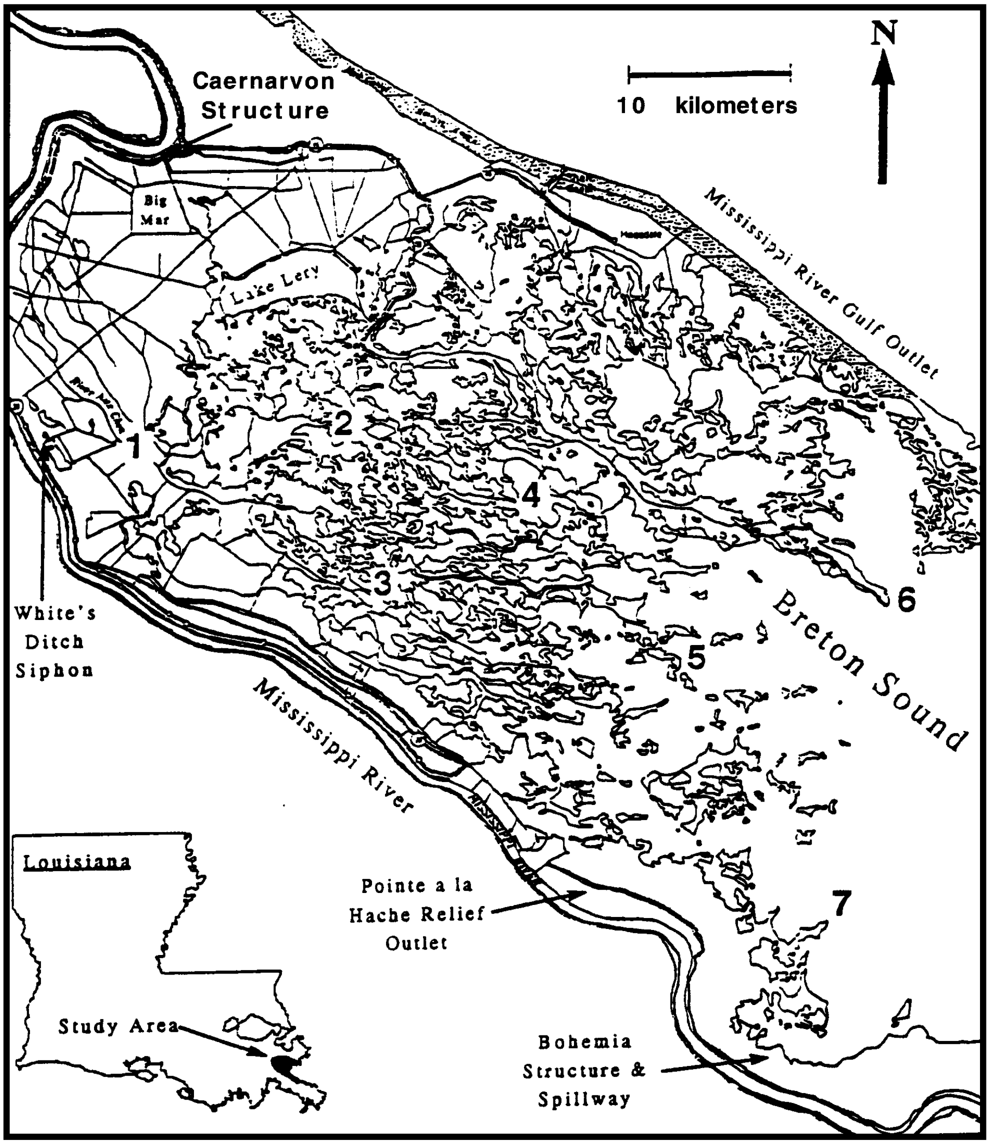


FIGURE 3.23. Caernarvon diversion and Breton Sound Estuary in coastal Louisiana. NOTE: Numbers refer to water quality monitoring stations. The Caernarvon, White's Ditch Siphon, and Bohemia structures are controlled freshwater diversions, and the Pointe a la Hache relief outlet and Bohemia spillway are areas of seasonal flooding of the Mississippi River.

Lane and Day (1999) analyzed nutrient data taken before and after the diversion was opened in 1991. A Before-After, Control-Impact (BACI) analysis, along with station-by-station (station numbers in Figure 3.23) contrasts, indicated that the diversion created no significant difference in nitrates or ammonium in the wetland and coastal waters but that total Kjeldahl nitrogen (TKN) and total nitrogen (TN) levels at stations 2, 5, and 6 decreased significantly as a result (Figure 3.24). Station 7 had high nitrate and total nitrogen concentrations compared to the other marsh stations due to the addition of river water in the region by the Bohemia structure and spillway. Mean pre- and post-diversion Mississippi River water nitrate concentrations ranged from 1.2 to 1.6 mg-N/L, while the marsh water quality ranged from 0.1 to 0.5 mg-N/L, suggesting rapid reduction in nitrate levels as river water entered the estuary. When the post-diversion data was broken down by season, nitrite + nitrate concentrations during summer and

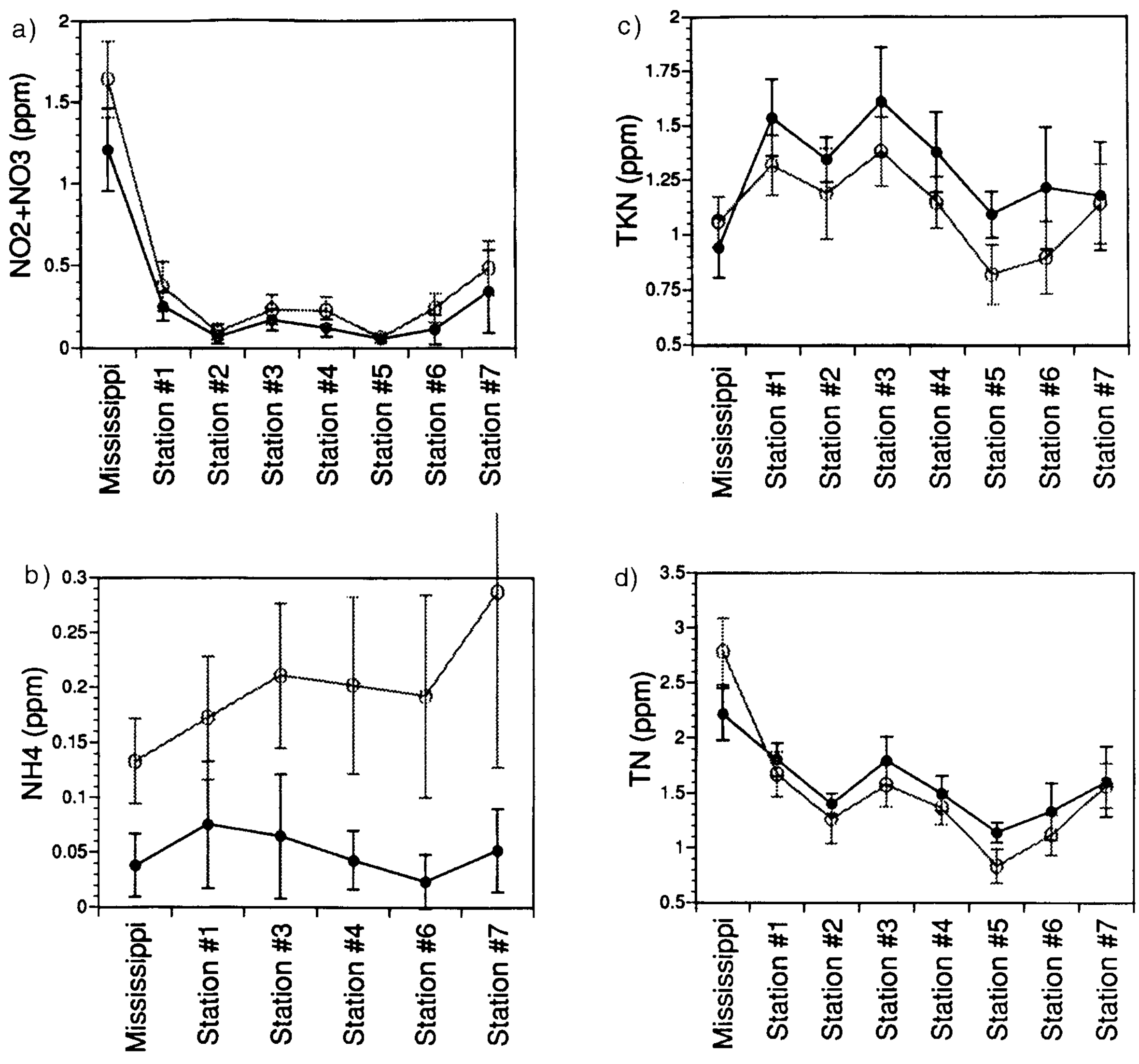


FIGURE 3.24. Pre- and post-diversion data with standard error bars for Louisiana's Breton Sound Estuary for (a) nitrite + nitrate, (b) ammonium–nitrogen, (c) total Kjeldahl nitrogen, and (d) total nitrogen. NOTE: Solid black data points are pre-diversion; open data points are post-diversion.

autumn were somewhat lower compared to winter and spring, but were not significantly different. Winter had the highest ammonium–nitrogen concentrations compared to the other seasons. Most important, salinity mixing diagrams (Figure 3.25; see Liss 1976 and Day et al. 1989 for a discussion of mixing diagrams) indicated that the Breton Sound system was acting as a strong sink for nitrite + nitrate and total nitrogen and as a source for ammonium–nitrogen and organic nitrogen. Salinity was considerably reduced at stations 1 through 4, indicating that the diversion significantly diluted estuarine water in the area.

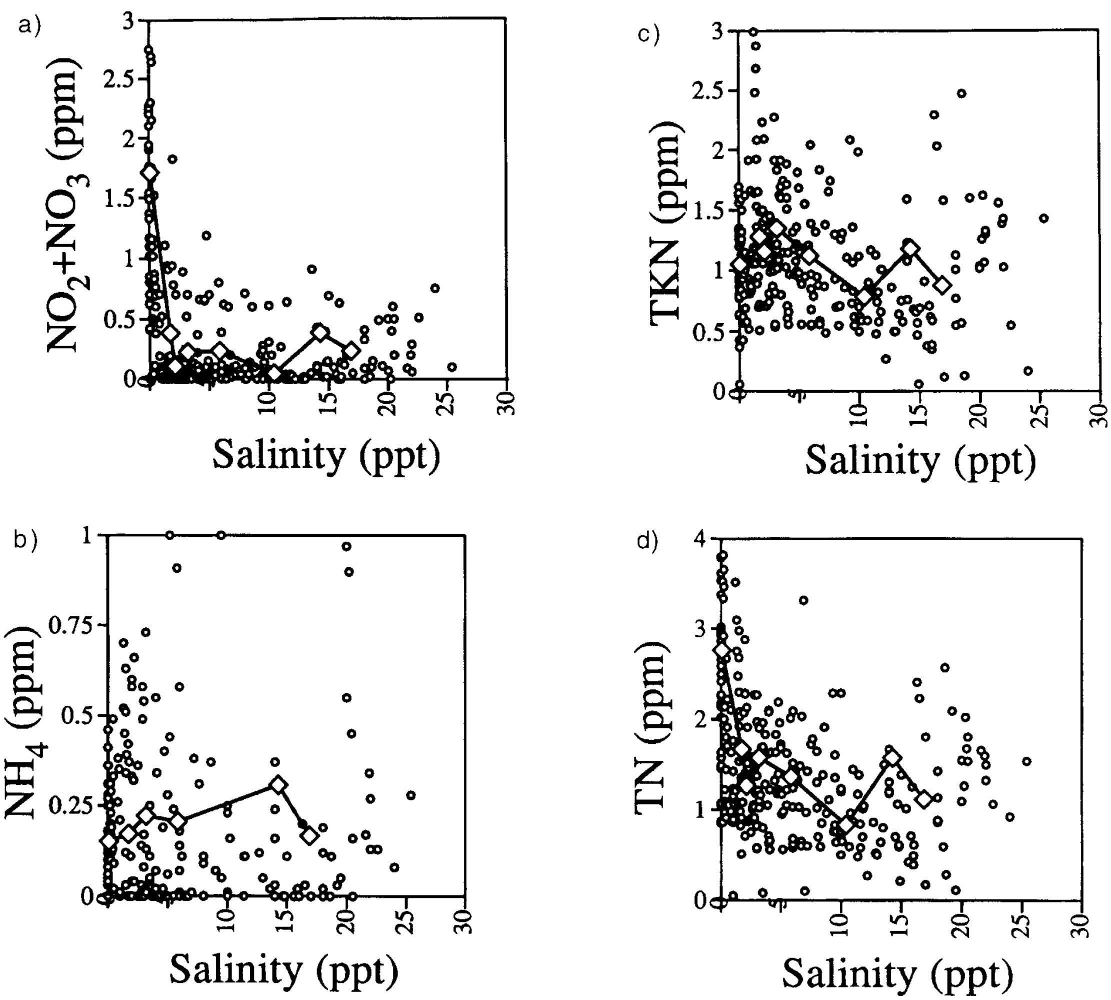


FIGURE 3.25. Post-diversion salinity mixing curve (with raw data points shown as circles) **and overall averages at each water quality station** (shown as diamonds) **for (a) nitrite + nitrate–nitrogen, (b) ammonium-nitrogen, (c) total Kjeldahl nitrogen, and (d) total nitrogen.**

Results of analysis of nutrient loading rates indicate 5.6–13.4 g-N m^{-2} yr^{-1} of nitrate–nitrogen, 8.9–23.4 g-N m^{-2} yr^{-1} of total nitrogen, and 0.9–2.0 g-P m^{-2} yr^{-1} of total phosphorus were delivered to the region north of the first two water quality monitoring stations during 1992–94 (Table 3.26). Removal efficiencies were 88–97% for nitrite + nitrate, 32–57% for total nitrogen, and 0–46% for total phosphorus.

TABLE 3.26. Nutrient loading rates and removal efficiency of wetlands north of the first two water quality monitoring stations at the Caernarvon freshwater diversion of the Mississippi River, Louisiana.

Parameters	1992	1993	1994
$NO_3 + NO_2$			
Loading *(g-N m^{-2} yr^{-1})*	5.6	7.3	13.4
Removal *(g-N m^{-2} yr^{-1})*	5.4	6.9	11.8
% Removal	97.0	95.0	88.0
Total Nitrogen			
Loading *(g-N m^{-2} yr^{-1})*	8.9	12.1	23.4
Removal *(g- N m^{-2} yr^{-1})*	5.1	5.7	7.5
% Removal	57.0	47.0	32.0
Total Phosphorus			
Loading *(g-P m^{-2} yr^{-1})*	0.9	1.4	2.0
Removal *(g-P m^{-2} yr^{-1})*	0.4	0.5	0.0
% Removal	46.0	39.0	0.0

3.6.3.2 THE BONNET CARRÉ SPILLWAY

The Bonnet Carré Spillway is a floodway designed to carry flood waters from the Mississippi to Lake Pontchartrain when New Orleans is threatened by high water levels (Figure 3.26). The spillway was constructed in 1931 after the devastating flood of 1927 and has been opened eight times since during high-water events. It is located 25 km upstream of New Orleans in an area of natural crevasses that breached the Mississippi River levee in the 1800s and introduced up to 4,000 m^3 sec^{-1} of water into Lake Pontchartrain (Davis 1993). The present spillway is designed to divert up to 7,000 m^3 sec^{-1} from the river during floods. The forested wetlands in the Bonnet Carré Spillway have shown no land loss and are healthy compared to swamp and marsh areas just upriver and downriver where there has been land loss.

Lake Pontchartrain is a large (1,630 km^2) oligohaline lake located in southeastern Louisiana, with a mean depth of about 3.7 m. In its natural state, the lake was surrounded by extensive wetlands, but large areas have been reclaimed or impounded on the south shore due to growth in the New Orleans metropolitan area. The lake receives freshwater input from several rivers as well as periodic openings of the Bonnet Carré Spillway, and is connected to the larger estuarine system through three large inlets (Figure 3.26). Two natural inlets—The Rigolettes and Chef Menteur Pass—communicate with Lake Borgne and Mississippi Sound; while a dredged canal, the Inner Harbor Navigation Canal (IHNC) is connected to Breton Sound. The Rigolettes, Chef Menteur, and the IHNC carry about 60%, 30%, and 10%, respectively, of the tidal exchange, and maximum combined flow is about 6,400 m^3 sec^{-1} (Swenson and Chuang 1983).

The spillway was opened in 1997 during the fourth-largest flood of the century (Day et al. 1999). Water flow through the structure increased to 6,800 m^3 sec^{-1} (240,000 cfs, or about 16.4% of the total flow of the river) on March 25–26, and then gradually declined due to both decreasing river discharge and closure of the structure. Concentrations of the different parameters in Mississippi River water varied during the diversion; TKN was 0.34–0.93 mg-N/l; total phosphorus, 0.17–0.33 mg-P/l; ammonium–nitrogen, 0.08–1.26 mg-N/l; nitrate–nitrogen, 1.08–1.26 mg-N/l; and total suspended solids, 34–110 mg/l. The introduction of river water reduced salinity and increased nutrient levels in the lake. This condition was most pronounced at stations 2 and 3, where the system went completely fresh within two weeks of the opening and nutrient levels were the same as those in the river, indicating essentially unchanged river water. At these two stations, high concentrations of nitrate and total phosphorus persisted for about a month after the closure of the structure but reached pre-opening levels by early to mid-June. By contrast to the rapid declines in nutrient concentration, salinity gradually increased, reaching pre-opening levels by late August/early September.

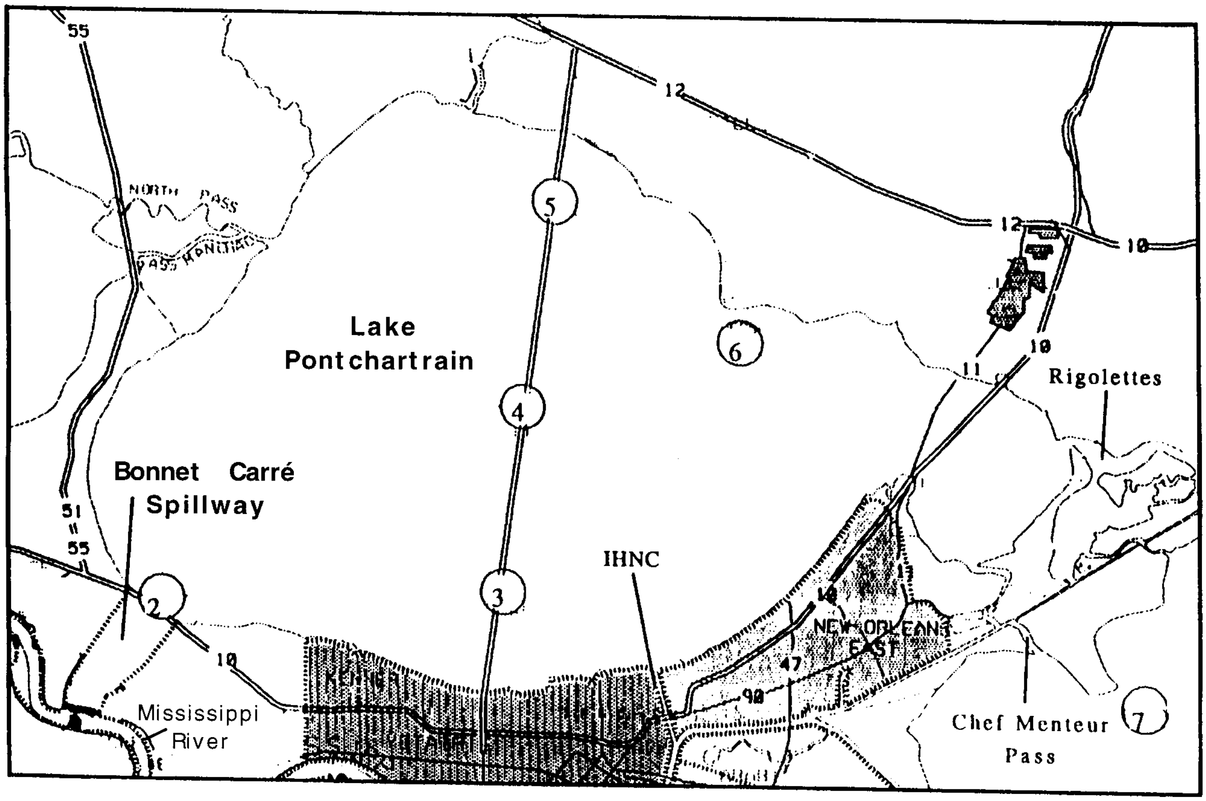


FIGURE 3.26. Map of Lake Pontchartrain near New Orleans, Louisiana, showing the Bonnet Carré Spillway and sampling stations.

At Station 4 in the mid-lake, nitrate and total phosphorus concentrations reached the levels in the river, but a week or two later than at Station 2. Salinity declined more slowly and never reached completely fresh conditions. As with Stations 2 and 3, nutrient concentrations returned to pre-opening levels by mid June, while salinity did not return to pre-opening levels until September. The north shore (Station 5) was less effected by the river inflow, and nutrient concentrations were lower. There was less influence of river water at Station 6 in the northeastern portion of the lake than at Station 7. Nitrate and total phosphorus concentrations at Station 6 were generally less than half of that in river water, while concentrations at Station 7 were close to river water for much of April and early May. Concentrations returned to pre-opening levels by mid June. Salinity was near fresh at Station 7 for about two weeks in mid April, while salinity was relatively higher at Station 6.

Following the closure of the spillway, there was an extensive blue-green algal bloom, predominantly *Anabaena circinalis* and *Microcystis aeruginosa*, in Lake Pontchartrain from late May that persisted through July (Dortch et al. 1998; Porrier and King 1998). Both of these species are capable of positive buoyancy, which allows them to avoid light limitation in turbid waters, and are known to be stimulated by excess nutrients (Dortch et al. 1998). Fish kills attributed to the bloom were reported during June and July, when algal cell counts were as high as 10^{10} cells per liter (Porrier and King 1998). Organic nitrogen levels in the central and northern portions of the lake increased during the period, the algal bloom was observed.

The introduction of fresh water in estuaries has been found to have broad effects on phytoplankton productivity. High primary productivity in estuaries receiving fresh water has been related to the introduction of nutrients (Nixon 1981), but production is also limited by light availability, which is attenuated by high suspended sediment concentrations usually associated with freshwater inputs (Cole and Cloern 1984). In river-dominated estuaries—environments with suspended solid concentrations often exceeding 50 mg/L—light is attenuated rapidly in the water column and phytoplankton photosynthesis is confined to a shallow photic zone. For this reason, phytoplankton productivity in turbid estuaries is often higher in the coastal ocean, adjacent to estuaries, where suspended sediment has dropped out of the water column yet high nutrient concentrations are still available (Cloern 1996). High chlorophyll concentrations after the spillway was closed were probably due to this effect, with lower sediment concentrations in the lake due to settling, yet high residual nutrient availability.

Clearly, the Bonnet Carré diversion had both positive and negative effects. There was a significant reduction of nitrate as the water flowed through the lake. There was also a large algal bloom. Diversions should be carried out in a way that maximizes benefits, such as flood control and nutrient retention, while reducing detrimental impacts, such as algal blooms.

3.6.3.3 The Atchafalaya Delta Region

About one-third of the Lower Mississippi River is discharged via the Atchafalaya River (Figure 3.27); the Atchafalaya Delta region, therefore, has a strong riverine influence. This area has the lowest land-loss rates in the Louisiana coastal zone (Britsch and Dunbar 1993), and two new deltas are forming in Atchafalaya Bay. In addition, the accretion and elevation gain offsets relative sea level rise in marshes surrounding Atchafalaya Bay (Baumann et al. 1984; Cahoon et al. 1995). Riverine input to the area has maintained high-elevation marshes that drain well and are characterized by strong elevation gains and high soil strength (Kemp et al. in press). This beneficial effect extends from fresh to saline marshes.

A number of studies of the water chemistry in the area have been conducted, especially in Fourleague Bay, which is strongly affected by Atchafalaya River discharge. Fourleague Bay is a shallow (mean depth 1.5 m), highly turbid, vertically well-mixed estuary surrounded by extensive fresh, brackish, and saline wetlands. The Atchafalaya River's discharge strongly affects the bay. Salinity ranges from 0–8 ppt in the upper bay to 0–26 ppt in the lower bay (Caffrey and Day 1986; Madden et al. 1988). During peak spring river discharge, the bay can be flushed in as little as seven days, while during low-discharge periods the residence time of the bay increases to about 65 days (Madden et al. 1988). Fourleague Bay is connected to the Gulf of Mexico via Oyster Bayou, a 4-km-long tidal channel. The bayou is the only direct outlet of Fourleague Bay, and peak current velocities can exceed 2.0 m sec^{-1}. Extensive intertidal and subtidal oyster reefs line the bayou, which is bordered by *Spartina alterniflora* marshes.

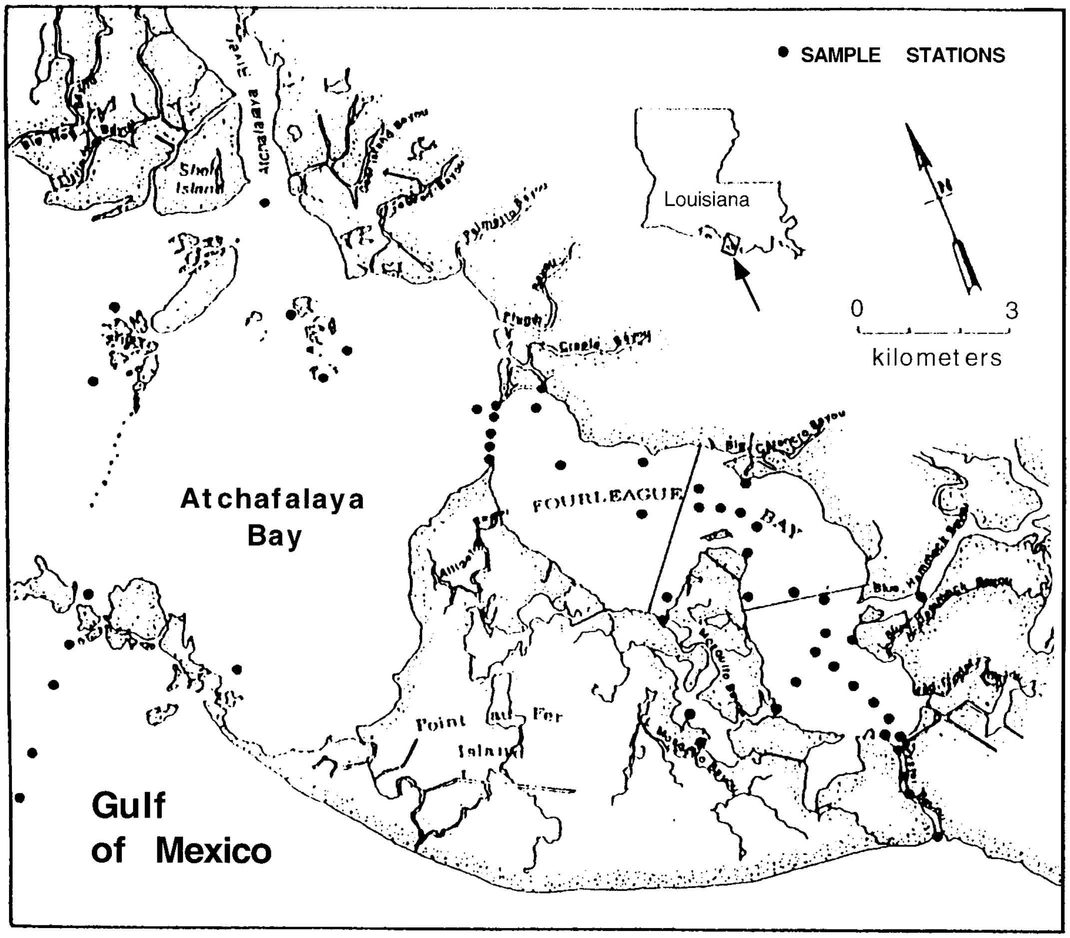
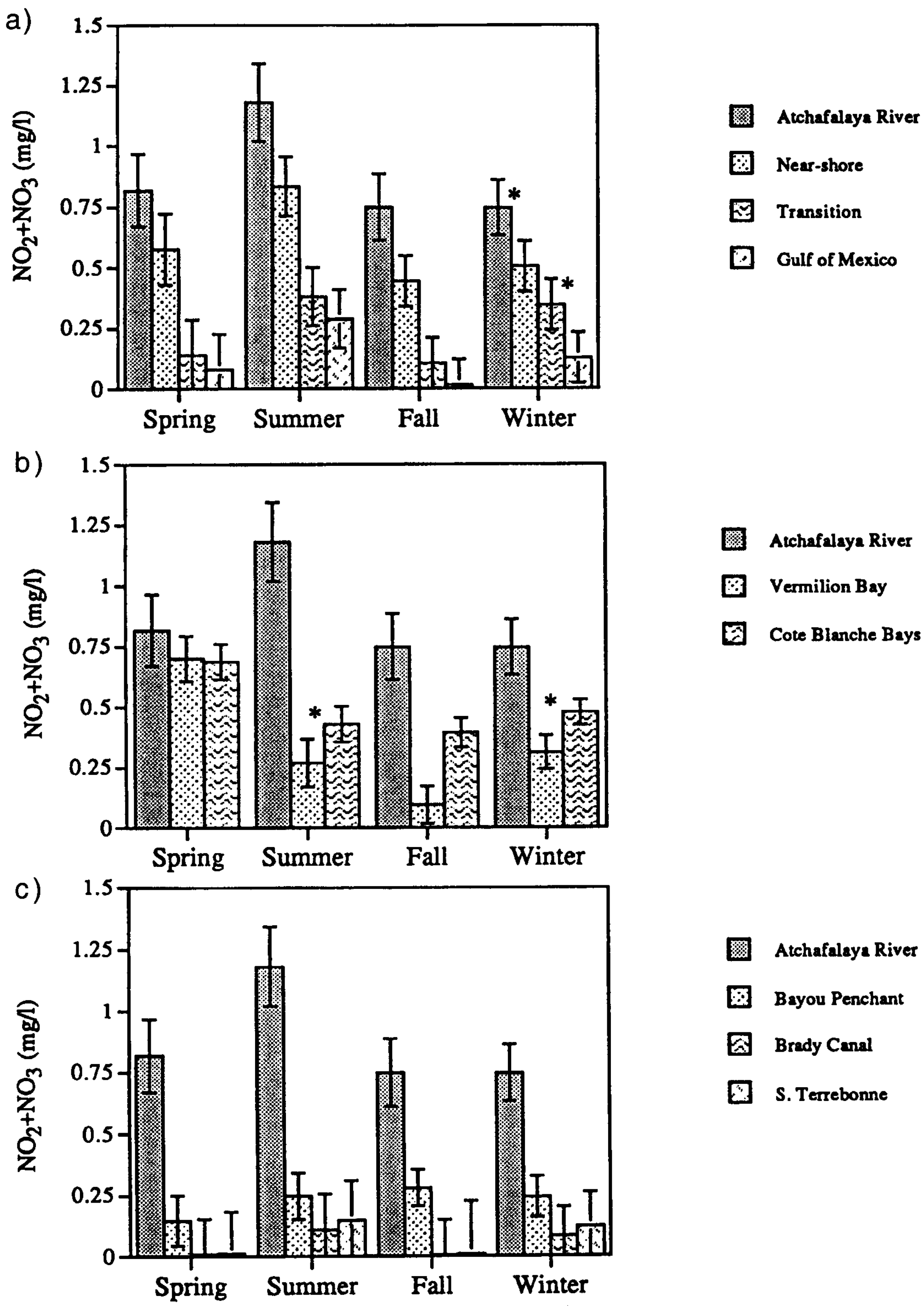


FIGURE 3.27. Atchafalaya Bay and Fourleague Bay in coastal Louisiana.

Nitrate concentrations are higher during the spring flood months between December and June, averaging 0.7–1.0 mg-N/L. From July to November, mean nitrate is 0.1–0.5 mg-N/L (Figure 3.28). Concentrations are significantly lower in the lower bay. During most of the year there was a strong nonconservative uptake of nitrate. Ammonium is seasonally less than 0.1 mg-N/L and does not vary much. As a result of high nitrogen inputs in the winter and spring and relatively stable phosphorus concentrations, the N:P ratio in all of the bay was greater than 16:1, indicating potential phosphorus limitation. Ratios were especially high in the upper and middle parts of the bay, exceeding 60:1 much of the time. In summer, as nitrogen inputs declined, the bay shifted from phosphorus to nitrogen limitation as ratios fell to below 10:1 in the middle and lower sections.

Mixing diagrams indicate that the river is always the primary source of nitrate to the bay and that the bay is nearly always a strong sink for nitrate. Evidence for dilution was found in March and April. Mixing diagrams for ammonium most often increased in the bay, indicating that the bay was a source for this compound. The reduction in nitrate concentrations, however, was almost ten times the increase in ammonia. Nitrate levels were also measured in water flowing from the Atchafalaya River to the Gulf of Mexico. Figure 3.28 illustrates significant reductions in nitrate concentrations from the Atchafalaya to the nearshore Gulf.



3.28. Nitrate concentrations in various transects from Atchafalaya River to coastal waters for spring, summer, fall, and winter. NOTE: Error bars indicate 95% confidence intervals of the mean. Regions with non-overlapping error bars are significantly different. An asterisk (*) indicates a significant difference for regions with questionable overlapping error bars (a < 0.0125).

3.6.4 Advantages and Limitations of Coastal Restoration

The information presented above suggests that diversion of river water into the coastal zone can have two beneficial effects. First, diversions can help reduce the rate of land loss and create new land. Land loss is low in the Atchafalaya and Caernarvon outfall areas and in the Bonnet Carré Spillway, and new land has been created in the Atchafalaya Delta and at Caernarvon. Second, some quantities of nitrate can be removed, thus reducing the input to the nearshore zone. The amount of potential nitrate reduction is probably limited to less than 10–15% of total flux in the river.

Flow of water through inshore areas should result in a strong reduction in the N:P and Si:N ratios. It is likely that similar changes will also occur in the ratios of N:Si, thus creating conditions more favorable for diatom growth, as described by Justićet al. (1995). Data from the different diversions indicate that the N:P and Si:N ratios are reduced. There seems to be a consensus among marine scientists that lowering the Si:P and Si:N ratios with respect to nutrient requirements of diatoms (~Si:P = 16:1 and Si:N = 1:1, for nutrient replete diatoms) will most likely increase the incidence of nonsiliceous blooms in the coastal waters. These blooms often include noxious and toxic forms. Turner et al. (1998) indicated that "diatom growth becomes Si-limited when the atomic ratio of silicate to dissolved inorganic nitrogen (Si:DIN) approaches 1:1." There is good evidence on this from a variety of coastal and estuarine waters that is summarized in Smayda (1989, 1990), Officer and Ryther (1980), Conley et al. (1993), and Justićet al. (1995).

The effectiveness of these large-scale diversions on significant uptake of nitrate–nitrogen has been controversial and is not universally accepted (Turner 1998). Therefore, additional studies are needed to determine the actual discharge rates of diversions, the area of wetlands needed, potential and actual nitrogen reductions, and the linkages between riverine input and offshore response. Although a 10–15% reduction of NO_3 loading during the spring flood may have significant beneficial impacts on offshore production and hypoxia, several uncertainties remain. Without upstream controls, the deltaic system may become nitrogen-saturated or it may release nitrogen in a form and season different from when it entered the delta.

3.7 UPPER MISSISSIPPI RIVER FLOOD CONTROL AND RESTORATION

Controlling floods in the Mississippi River Basin by restoring and creating wetland areas was much discussed after the disastrous flood of 1993. A number of benefits are associated with wetland restoration along the Mississippi River and its many tributaries, such as enhanced wildlife habitat and recreational areas and, on a more practical note, enhanced flood control. In 1993, the flood waters that devastated significant parts of the 1.8 million km^2 of the Upper Mississippi Basin could have been partly held within an estimated 5.3 million hectares (13 million acres) of wetlands (Hey and Philippi 1995). Most important, related to the subject of this report, if these wetlands had been able to capture and store the flood waters, the emanating discharge would have contained far less nitrate–nitrogen. Fewer wetlands would be needed to treat the mean annual flood or, even the smaller, mean annual flow. Flood control in the Upper Mississippi would also decrease the necessity or the possible overload (hydrologically and chemically) of downstream Mississippi River diversions.

During the Upper Mississippi River flood of 1993 flood waters rushed southward through channels and over floodplains incapable of storing flood waters without economic loss or incapable of supporting the anaerobic environments necessary for the nitrate removal. Consequently, $16 billion in losses resulted and large nitrate loads were swept downstream into the Gulf of Mexico. Applying the same criteria as used for the Upper Mississippi Basin to the larger basin at Vicksburg, Mississippi, of 2.95 million km^2, approximately 10 million hectares (25 million acres) of wetlands would be necessary to effectively reduce the peak flow. On the other hand, only 6 million hectares (15 million acres) would be required to reduce the nitrate load of the mean annual flow by about 80%. The larger area represents less than 3% of the basin, and the smaller less than 2%.

CHAPTER 4

Reducing Nutrient Loadings to the Gulf of Mexico

4.1 BEST PRACTICES FOR REDUCING NITROGEN LOADINGS

Of the methods reviewed in Chapter 3, the most appropriate for reducing nitrogen that would otherwise reach the Gulf of Mexico appear to be the following:

1. Change farming practices to minimize nitrate loss by reducing the use of nitrogen, principally by decreasing the use of excess nitrogen fertilizer, enhancing the use of manure nitrogen, and applying an array of best management practices on farms.
2. Intercept laterally moving ground water and surface water from farmland with riparian zones and created and restored wetlands, particularly targeting areas with artificial subsurface drainage and high concentrations of nitrates.
3. Install tertiary treatment systems, particularly treatment wetlands, for the removal of nitrate on major sources of domestic wastewater in the Mississippi River Basin.
4. Provide a system of river-diversion backwaters in the Mississippi River Delta, particularly for intercepting large fluxes of nitrogen associated with floods, and river flood backwaters in the Upper Mississippi for both flood control and nitrogen retention.

Each of these management practices, presented in Table 4.1, has potential as well as limitations for use in selected areas. Although no single practice is applicable everywhere, the authors believe that one practice can reduce nitrogen entry into surface waters from the vast majority of the Mississippi River Basin. The challenge lies in selecting and adapting the most appropriate approach to fit local landscape conditions.

This chapter presents the specifics of the four alternatives, along with rough estimates of the possible decreases from applying each of them. Estimated decreases in sources and in the outputs to the Gulf of Mexico are based on data in Table 1.1, which suggest that total nitrogen loadings from various sources to the Mississippi River Basin are on the order of 20 million metric tons/year, but only about 1.6 million metric tons/year of nitrogen are measured near the discharge of the Mississippi River to the Gulf of Mexico (Goolsby et al. 1999).

4.2 CHANGING FARM PRACTICES

Variable factors, such as climate, have a profound influence on nitrate–N concentrations and loadings in subsurface drainage water. The dynamics of nitrogen behavior in drained agricultural soils during periodic climatic events, particularly wet years, and the management of both crops and nutrient inputs (controllable factors) must be considered carefully by agriculturalists as they manage the land. These factors must be understood by scientists and policymakers as they edu-

TABLE 4.1. Recommended approaches for reducing significant amounts of nitrogen (N) loading to streams and rivers in the Mississippi River Basin and the Gulf of Mexico.

Approach	**Potential N Reduction** ***10^3 metric tons/yr***
Change Farm Practices[1]	
N-management—Reduce "insurance" rates of N fertilizer application, properly distribute manure, apply appropriate credits for previous crop legumes and manure, along with improved soil N testing methods.	900–1,400
Alternative cropping systems—Substitute perennial crops for 10% of the present corn–soybean area.	500
Create and Restore Wetlands and Riparian Buffers	
Create and restore 21,000–53,000 km^2 (5–13 million acres) of wetlands in the Mississippi River Basin (0.7–1.8% of the basin).	300–800
Improve management of animal manure in livestock-production areas.	500
Limit minimum spacing between farm drainage tiles to 15 meters.	?
Restore 78,000–200,000 km^2 (19–48 million acres) of riparian bottomland hardwood forest (2.7–7.0% of the basin).	300–800
Reduce Point-Source Pollution	
Apply tertiary treatment to domestic wastewater.	20
Control Flooding in the Mississippi	
Divert river flow in the Louisiana delta.	50–100

[1]Estimated on-site source reductions do not translate to equivalent reductions in Gulf of Mexico nitrogen loading, as only about 8% of nitrogen sources reach the lower Mississippi River (see Table 1.1).

cate the public and develop environmental guidelines regarding the loading of nitrates to surface waters. Several specific farm practices need to be undertaken:

1. Nitrogen management, including both fertilizer and manure application, needs to be fine-tuned on the farm. This responsibility rests with the farmer, the supplier of the nutrients (fertilizer dealer or manure supplier), and the crop consultants. Applying the correct rate of nitrogen at the optimum time has been shown to substantially decrease nitrate losses over excessive nitrogen application during the wrong periods. Knowing the nutrient content and application rate of the manure, spreading it uniformly, and incorporating it in a timely manner will lead to better management and confidence in manure nitrogen as a nutrient source. If nitrogen were better managed on farms in the Mississippi River Basin by reducing "insurance" rates of nitrogen fertilizer application, properly distributing manure, and applying appropriate credits for previous legume crops and animal manure applications, nitrogen sources to streams and rivers via subsurface drainage could be reduced by about 10–15%, or about 0.9–1.4 million metric tons/yr.
2. Continued development and application of improved soil N-testing methods to determine the availability of mineralizable and carryover N from the previous crop would be helpful, especially following dry years, legumes, or past manure applications. Pre-planting and in-season soil nitrogen tests have been developed in the past 10–15 years to help farmers arrive at better rates of nitrogen application by assessing available nitrate in the soil. Use of these tests should greatly help prevent overapplication of N fertilizer when residual nitrate remains from the previous crop, which will most likely happen following a dry year, when manure was applied to the previous crop and when corn is the crop.
3. Alternative cropping systems that contain perennial crops would greatly reduce nitrate losses. Obtaining a market and a satisfactory economic return are obstacles farmers currently face. If alternative cropping systems involving perennial crops, such as alfalfa or grass–alfalfa mixes, were

substituted for some of the present corn–soybean areas in the Mississippi River Basin, nitrate losses would decrease by about 90% in these areas. Substituting perennial crops in 10% of the corn–soybean farms in the basin could decrease loadings to streams and rivers by an estimated 0.5 million metric tons/year

4. Improved management of animal manure and subsequent runoff in livestock-producing areas would help lower nitrogen losses substantially. Decreasing feedlot runoff by 20% could reduce runoff to streams and rivers in the basin by 0.5 million metric tons/year.
5. Although narrowing the spacing between tile lines may be desirable to many crop producers because of quicker drainage of excess water from the soil profile, keeping the spacing at > 15 m would likely be a good compromise that would allow adequate drainage without escalating nitrate losses in the subsurface drainage water.

4.3 INTERCEPTING AGRICULTURAL DRAINAGE WITH WETLANDS AND RIPARIAN BUFFERS

Wetlands, riparian zones, and controlled drainage projects share a common feature in that they are viable alternatives for serving as buffers between agricultural uplands and streams and rivers. Furthermore, all can be designed in the landscape to enhance nitrate–nitrogen reduction through two main ecological processes—denitrification and nitrogen uptake by plants (the latter is important only if nitrogen is stored in soil/biomass for a long time or is harvested and taken out of the basin).

Studies cited above suggest ecological engineering design parameters of nitrogen reduction of about 4 g-N m^{-2} yr^{-1} with riparian forests and about 10–20 g-N m^{-2} yr^{-1} with restored/created wetlands. Thus, wetlands appear to be 2.5–5.0 times more “efficient” in reducing nitrogen loading per unit area. These design criteria were used to estimate the general extent of wetlands and riparian zones necessary to significantly reduce the nitrogen, particularly nitrate–nitrogen, reaching streams and rivers in the Mississippi River Basin.

4.3.1 Wetlands

Approximately 21,000–52,000 km^2 (5–13 million acres) of newly created and restored wetlands in the Mississippi River Basin (0.7–1.8% of the basin) would be needed to reduce nitrogen loading to the Gulf of Mexico by 20–50% (Tables 4.1 and 4.2). To put this in perspective, enforcement of Section 404 of the Clean Water Act through wetland mitigation has resulted in an estimated net gain of 336 km^2 (83,000 acres) of wetlands in the entire United States (Mike Davis, personal communication), while conservation easement practices and partners-in-wildlife programs in the upper Midwest have increased wetland area by approximately 600 km^2 (150,000 acres). The national conservation set-aside programs in agriculture, particularly the Conservation Reserve Program (CRP), were estimated to restore about 360 km^2 in the United States from 1987 to 1990 (Mitsch and Gosselink 1993), and the North-Central U.S. office of the U.S. Fish and Wildlife Service estimated it had restored approximately 280 km^2 (70,000 acres) in the upper Midwest through 1997 (K. Kroonmeyer, personal communication). An effort estimated to be 20–50 times current efforts of wetland restoration and creation would be needed in the Mississippi River Basin to achieve the goal of 21,000–52,000 km^2 of restored wetlands.

The most effective use of wetland restoration and creation would be in watersheds that discharge high amounts of nitrogen (Table 4.2). For example, the Illinois River Basin, with 7% of its watershed converted to wetland, could reduce about 50% of the 144,000 metric tons/yr of nutrients it generates, or about 5% of the entire nitrogen load to the Gulf of Mexico. In contrast, the James River Basin in South Dakota contributes only 1,178 metric tons/yr of nitrogen to the basin; controlling all of this discharge would retain only 0.075% of the nitrogen load to the Gulf. So clearly, restoring and creating wetlands in the Mississippi River Basin would have to be strategic; that is, wetlands should be located where agricultural sources of nitrogen and subsurface drainage are the largest. A reasonable strategy for the Mississippi River Basin is to strategically point the conservation easement program toward conserving the Gulf’s ecology by restoring flooding to lands that have been drained and are now exporting excessive amounts of nitrate–nitrogen to the Mississippi River Basin.

TABLE 4.2. Estimated area of riparian forests needed to control nitrogen in the Mississippi River Basin (MRB) and selected sub-basins, assuming a reduction of 15 g-N/m²-yr.

River Basin	Drainage Area *(km²)*	Discharge *(metric tons-N/yr)*	Total Wetland Area *(km²)* Required for Specific % Nitrogen Reductions			% of Watershed Required for Specific % Nitrogen Reductions		
			70%	*50%*	*20%*	*70%*	*50%*	*20%*
Entire MRB	2,953,895	1,568,000	73,173	52,267	20,907	2.5	1.8	0.71
Raccoon R., IA	8,912	27,520	1,284	917	367	14.4	10.3	4.12
Illinois R., IL	68,800	144,320	6,735	4,811	1,924	9.8	7.0	2.80
Scioto R., OH	13,289	23,330	1,089	778	311	8.2	5.9	2.34
Osage R., MI	37,555	15,410	719	514	205	1.9	1.4	0.55
James R., SD	55,814	1,170	55	39	16	0.1	0.07	0.03

Source: Nitrogen discharge from Goolsby et al. 1999.

4.3.2 Riparian Zones

Decreasing the load of nitrogen to the Mississippi River Basin through riparian zones also appears to be a viable alternative, although the amount of denitrification and, hence, nitrogen-uptake in these systems appears from the literature to be less effective per unit area than wetlands. Using an analysis similar to the one above, an estimated 3–7% of the Mississippi River Basin, or 78,000–200,000 km² (19–48 million acres) would have to be restored to riparian bottomland hardwood forest (and/or controlled drainage) to control 20–50% of the nitrogen reaching the Gulf (Tables 4.1 and 4.3). Because most of these riparian forests would export organic nitrogen to the streams and rivers as litterfall, the estimate is probably optimistic. Furthermore, to be effective, these restored bottomland riparian systems would have to located near the major sources of subsurface nitrate drainage.

To put these numbers in context, there are currently an estimated 120,000 km² of forested wetlands (mostly as riparian forests) in the North Central and South Central regions of the United States. About 50,000 km² are in the North Central part of the Mississippi River Basin, where most of the serious nitrogen sources to the MRB exist. A concerted effort to double or triple the amount of riparian forests and similar buffer systems in the basin would be necessary to have a significant impact on the nitrogen load of the basin to the Gulf of Mexico.

TABLE 4.3. Estimated area of riparian forests needed to control nitrogen in the Mississippi River Basin (MRB) and selected sub-basins, assuming a reduction of 4 g-N/m²-yr.

River Basin	Drainage Area *(km²)*	Discharge *(metric tons-N/yr)*	Total Wetland Area *(km²)* Required for Specific % Nitrogen Reductions			% of Watershed Required for Specific % Nitrogen Reductions		
			70%	*50%*	*20%*	*70%*	*50%*	*20%*
Entire MRB	2,953,895	1,568,000	274,400	196,000	78,400	9.3	6.6	2.7
Raccoon R., IA	8,912	27,520	4,816	3,440	1,376	54.0	38.6	15.4
Illinois R., IL	68,800	144,320	25,256	18,040	7,216	36.7	26.2	10.5
Scioto R., OH	13,289	23,330	4,083	2,916	1,167	30.7	21.9	8.8
Osage R., MI	37,555	15,410	2,697	1,926	771	7.2	5.1	2.1
James R., SD	55,814	1,170	205	146	59	0.4	0.3	0.1

Source: Nitrogen discharge from Goolsby et al. 1999.

4.3.3 Local Benefits

Constructing and restoring wetlands and riparian zones in the Mississippi River Basin to control nonpoint-source pollution would contribute to several important national goals (in addition to reducing the hypoxia in the Gulf of Mexico), including cleaning up the waterways in the Midwest, particularly for drinking-water protection, adding to the nation's disappearing wetland habitat, improving river ecosystems, enhancing terrestrial wildlife in river corridors, and mitigating the effects of floods. These so-called "local benefits" would accrue to the local regions in which the wetlands and riparian zones are restored.

4.3.3.1 LOCAL WATER QUALITY IMPROVEMENT

Reducing nitrates would provide a much-needed improvement in stream and river water quality throughout the Midwest. The decrease in nitrate–nitrogen would result in safer drinking water for many communities in the Midwest and fewer nitrate alerts, the latter a common occurrence every spring in many parts of the Midwest. High concentrations of nitrates in drinking water cause a disease called methemoglominemia or "blue baby." As a result, water treatment plants in the Midwest watch nitrate concentrations in both ground water and surface water carefully, particularly in the spring. High levels of nitrates in streams can also contribute to eutrophication of inland waters when phosphorus is also abundant.

4.3.3.2 WETLAND RESTORATION

Restoring wetlands in the Mississippi River Basin is in keeping with an ambitious recommendation by the National Research Council's Committee on the Restoration of Aquatic Ecosystems (NRC 1992), which called for a national program of wetland restoration that would contribute to an overall gain of 10 million acres [4 million hectares] by the year 2010. Well-placed wetlands and riparian buffers generally support larger populations of wildlife because of the diverse habitats they provide. In wetlands with varying seasonal water depths and patterns of open water, emergent vegetation, and mud flats, a wide variety of birds, including herons, egrets, and rails, find their proper feeding niche. Amphibians, especially frogs, are often considered the "canary" of the landscape because of their susceptibility to pollution, but they can thrive in a well-functioning wetland.

Wetlands and riparian forests are among the most productive ecosystems in the landscape. This productivity is translated into the production of detritus and the support of food chains, both terrestrial and aquatic. The hydrologically open nature of these systems means that they are continually receiving propagules of plants, animals, and microbes from upstream systems. It also means that they are continually exporting organic carbon to downstream and/or adjacent aquatic systems. All of this biodiversity would occur wherever the wetland is built, not in a far-away coastal area; the habitat benefit would be local.

4.3.3.3 RIVER ECOLOGY ENHANCEMENT

Another recommendation by the Committee on the Restoration of Aquatic Ecosystems called for restoring 67,000 km (40,000 miles) of streams, rivers, and floodplains. The riparian zone restoration recommended in this report would make a major contribution toward that goal. The restoration of riparian vegetation contributes several advantages to the ecology of streams and rivers (USEPA 1995b). Roots of riparian vegetation stabilize the stream bank and prevent stream bank erosion and sedimentation. Stabilized stream banks also help maintain the geometry of the stream, including such characteristics as the meander length and profile. Tree roots and woody debris are also important habitat features for macroinvertebrates and fish. Overhanging stream banks, stabilized by tree roots and large woody debris, can be important habitat for fish. Large woody debris provides critical macroinvertebrate habitat and can also create dams and trap sediment and detritus. Streamside vegetation also affects the amount of sunlight that reaches the stream and, in turn, the temperature of the water.

4.3.3.4 TERRESTRIAL WILDLIFE ENHANCEMENT

Wildlife habitat is greatly enhanced whenever riparian wetlands are restored. In a stratified riparian forest, different habitat zones exist vertically, including the soil–air interface, herbs and shrubs, intermediate-height trees, and the canopy. Included with the leaf litter and rotting logs at the soil–water interface are insects, isopods, spiders, and mites. These organisms are a food source for reptiles, mice, and birds. The herbs and shrubs provide habitat for insects, birds, and mammals. The intermediate zone and the canopy serve as habitat for birds, bats, squirrels, opossums, and raccoons. Bird habitat may be highly stratified, and birds generally show a preference for certain layers that differ in habitat characteristics and food sources. Most important, wetlands and riparian forests serve as corridors linking dryer, less diverse uplands to more moist, more diverse bottomlands and are the natural highways for waterfowl and other birds, as well as numerous terrestrial animals.

4.3.3.5 FLOOD CONTROL

The role of floodplains and backwater wetlands in storing flood waters is an often overlooked value of these systems. The NRC (1992) reported that "their position in the landscape, whether as isolated wetlands or floodplains contiguous with rivers and streams, gives wetlands a major role in storage of floodwater and abatement of flooding." Hey and Philippi (1995) estimated that approximately 3% of the Upper Mississippi watershed, if restored back to backwaters and wetlands, would be sufficient to provide significant flood water retention even during the Upper Mississippi River flood of 1993. This is about 50,000 km^2 (13 million acres) of the Upper Mississippi River Basin. Interestingly, Hey and Philippi (1995) found that a similar area of restored wetland (53,660 km^2 or 13.3 million acres) would be sufficient to provide improved water quality, even during the 100-year flood in the Upper Mississippi River Basin.

Thus, combining the flood-control capabilities of wetlands with their function to retain nitrate makes using wetlands in the Upper Mississippi River Basin even more attractive. This goal of 3% of the watershed is only slightly higher than our estimated 1.8–2.5% of wetlands in the watershed necessary to significantly reduce nitrate–nitrogen in the river system. Wetland restoration on a large scale could thus protect the Gulf of Mexico during excessive flooding in two ways: (1) denitrification has a chance to occur in the flood water that the wetlands retain, and (2) the retention of upstream flood water prevents downstream systems from becoming overloaded.

4.4 TERTIARY TREATMENT OF DOMESTIC WASTEWATER

Both environmental technologies and ecotechnologies are available for controlling nitrogen in the Mississippi River Basin, particularly nitrate–nitrogen from domestic wastewater. Goolsby et al. (1999) estimated that domestic wastewater sources contribute 0.2 million metric tons of nitrogen/yr to the basin. If tertiary treatment, such as constructed wetlands and nitrification–denitrification basins, were used to control 50% of the nitrogen discharged from these point sources, the result would be a reduction of only a few percent of the load of nitrogen to the Gulf. Nevertheless, because nitrogen concentrations are relatively high and more easily controlled in point sources, requiring tertiary treatment for nitrogen in the basin should remain as a serious alternative. Wastewater wetlands offer the best alternative because of the lower costs and because of ancillary benefits, such as wildlife enhancement (Knight 1992) and others described above for nonpoint-source wetlands and riparian zones.

4.5 RIVER DIVERSIONS IN LOUISIANA

Some nitrate can be removed from the Mississippi's water if the river is diverted in large amounts over wetlands and shallow inshore waters in the Louisiana Delta, particularly during high flow. This assumes that the systems are not overloaded—i.e., the area is large enough, and the system does not become nitrogen-saturated with time. The issue is how much nitrate can be practically removed and whether it is enough to make any significant difference in offshore plankton production.

To estimate the potential for NO_3 removal, various portions of the total nitrogen entering the coastal zone through the Mississippi River were theoretically diverted to backwaters and adjacent wetlands in Louisiana. Using a retention rate of 10 g-N m^{-2} yr^{-1}, based on nitrate removal rates of 10 g m^{-2} yr^{-1} at Caernarvon, Louisiana, and on other studies documented in this report, a reduction of 50,000–100,000 metrics tons/yr of nitrogen could be achieved by diverting Mississippi River water in the delta region (Table 4.1). Removing 50,000 metric tons/yr would require about 500,000 hectares and diversion of about 13% of the total river flow. Removing 100,000 metric tons would require about 1,000,000 hectares and diverting about 26% of river flow. Because nitrate removal can take place in marshes, swamps, or shallow open water, these areas can be compared to the areas of these habitats in the coastal zone. The deltaic plain of the Mississippi Delta has at least 200,000 hectares of swamps, 1,200,000 hectares of marsh, and 1,400,000 hectares of inshore open water, or a total of 2,800,000 hectares of wetlands and shallow coastal areas.

4.6 MITIGATING ISSUES

4.6.1 Scale Effect

Caution is advised on applying any data derived in the small scale to entire watersheds. Of course, there have been no controlled studies at the scale of the Mississippi River Basin on the effects of management practices on actual retention of nutrients. There is some danger in extrapolating from small scale (e.g., 1–5 ha studies at the largest) to restoration that involves millions of hectares. Larger-scale studies appear to have less variability but are almost impossible to conduct in a controlled environment for a number of economic and institutional reasons.

4.6.2 Comparing "Apples and Oranges"

We have suggested a dual approach to on-site and off-site control of nitrogen as appropriate. But loss rates, such as those reported in Table 4.1, should be compared with caution. On-site source reduction due to such practices as reduction in fertilizer use do not translate to an equivalent reduction in load to the Gulf of Mexico. Table 1.1 illustrates that there are about 20 million metric tons of independent sources of nitrogen to the Mississippi River Basin (avoiding double counting) but only about 1.6 million metric tons of nitrogen reach the Gulf. Most of the difference is described by Goolsby et al. (1999) in food export and other losses.

Reducing nitrogen application on the farms by 1 million metric tons could cause a reduction as low as 0.08 million metric tons of nitrogen in the Gulf, using the ratio of 20:1.6. On the other hand wetlands and riparian zones intercept the drainage just before it reaches a stream or river. A reasonable assumption is that most of this nitrogen that leaves a farm field does reach the Gulf, as Goolsby et al. (1999) point out that there appears to be little in-stream loss of nitrogen once it reaches the streams and rivers. Reducing 1 million metric tons of nitrogen in a wetland conceivably causes a reduction of 1 million metric tons of nitrogen at the Gulf.

4.6.3 System Delay and Buffering

Two factors in the Mississippi River Basin confound the idea that a reduction of nutrients well up in the watershed will have an impact in the Gulf of Mexico. First there is a delay between the time that fertilizer and manure are applied and the time that nitrate appears in streams and rivers. Second, there is a considerable delay between the discharge of a kilogram of nitrogen in the upper part of the basin and its appearance in the Gulf. On its way, it has perhaps spiraled through the nitrogen cycle several times and is also subject to in-stream retention. On the other hand, if all sources of nitrogen were eliminated in the upper part of the basin, there would still be in-stream release of nitrogen chemicals from storages in the sediments of streams and rivers and from allochthonous sources along the streams—e.g. litterfall from riparian forests. It is almost impossible to estimate how important this buffering effect of in-stream processes would be.

4.6.4 Agricultural Production

Agricultural production would be minimally affected by some of these approaches, particularly on-farm practices, but there would be some loss of farmland with other practices, such as restored and created wetlands and riparian zones. If some of the nonagricultural alternatives given above, such as wetlands and riparian buffers, prove to be unacceptable or infeasible, major reductions in the use of nitrogen fertilizer may be the only way to significantly reduce concentrations of nitrates in streams and rivers in at least the northern half of the Mississippi River Basin. There are major agricultural policy implications of a significant reduction in nitrogen fertilizer use, but if the public is willing to pay for cleaner streams and rivers in the Midwest and reduced severity of the Gulf of Mexico hypoxia, reducing fertilizer use far in excess of levels projected here would need to be discussed.

4.6.5 Other Nutrients

The increase in nitrate–nitrogen observed in the Mississippi River near the Gulf over the past 50 years has not been a one-variable experiment. Sediment, phosphorus, and silicate loads have also changed as a result of pollution, dam building, and land-use change in the basin. Although a significant literature implicates nitrogen as the limiting factor in coastal waters around the world, there are confounding factors involved in determining if the reduction of a known amount of nitrogen will reduce the area of hypoxia. There is the question as to whether other chemicals, particularly phosphorus and silicate, are now co-limiting factors in the Gulf. As the amount of nitrogen has continued to rise in the Mississippi River, the N:P and N:Si ratios have increased to the point that phosphorus and silicate could be seasonally limiting.

4.6.6 Long-Term Prognosis

Even if we were able to reduce nitrogen loading to the Mississippi River by a substantial amount, there are no guarantees that this reduction would continue well into the future. Increases in populations in the basins, with their subsequent increased food requirements and domestic, commercial, and industrial waste production, would necessitate a continued increase in nitrogen control in the basin, or the system would slip back to loadings seen in earlier years.

4.6.7 Catastrophic Events

Catastrophic flooding, such as that seen in 1993 in the Upper Mississippi Basin, has a significant role in exporting a considerably greater amount of nitrogen to the Gulf and increasing the size of the hypoxia appreciably (Rabalais et al. 1998). Smaller scale studies of farmland have shown the same effect of wet years contributing considerably higher nitrogen concentrations to streams and rivers than do dry years (Randall, 1998). There is also the concern that catastrophic hydrologic events, such as floods, could overwhelm any engineered (ecological or otherwise) solution to nitrogen pollution. Therefore, ancillary benefits, such as flood control, that accompany approaches like wetland creation are really part of the overall solution, as they would reduce the significance of these catastrophic events that would otherwise overwhelm the control systems.

4.6.8 Uncertainty of Ecotechnology

Because of its scale, the ecological problem of Gulf of Mexico hypoxia cannot be solved with conventional technology alone. In fact, the costs would be overwhelming. So ecotechnology, the use of natural ecosystems to solve environmental problems, should be a prominent part of the solution. Since ecotechnological systems, by their nature, are not rigidly engineered systems, their performance has a wider variance than that of conventional engineered systems. This uncertainty must be factored into any expectation of immediate results. Riparian zones will grow up into large forests that will export high amounts of organic matter to streams and rivers, sending some of the captured nitrogen downstream. Wetlands could become saturated with nitrogen if most of the nitrate is not lost through denitrification. Overall performance of these systems could also be affected by wetland aging, excessive sedimentation, storms, and rivers changing courses over longer periods.

4.6.9 Production of Greenhouse Gases

The extensive development of wetlands in the Mississippi River Basin should lead to increased denitrification, which produces both N_2 and N_2O gases. In fact, denitrification is the primary process that needs to be accelerated in the Mississippi River Basin to reduce nitrate–nitrogen before it reaches the Gulf of Mexico. N_2O, a greenhouse gas, is considered 200 times more radiatively active than CO_2 on a molecular basis. It has also been increasing by 0.25% per year in the atmosphere (Rasmussen and Khalil 1986; Kang and Freeman 1998). Thus, consideration must be given to whether a massive increase in anaerobic zones (wetlands and riparian systems) in the Midwest would increase the emission of N_2O. While this question needs further research, there is some evidence that the increase in wetlands would not lead to any serious problems:

1. Most denitrified nitrogen is generally released as N_2 gas; the percentage that is emitted as N_2O gas decreases with decreased redox potential, lower nitrate concentrations, higher soil moisture, and higher pH (Weller et al. 1994). Wetlands have lower redox potential and higher soil moisture than do uplands. Restored and created wetlands would thus be expected to release less N_2O as a percentage of total N denitrified than more oxidized, drier soils.
2. N_2O is increasing annually in the atmosphere, despite a general decline in the extent of wetlands worldwide (Mitsch et al. 1994).
3. Agricultural fields are probably already significant sources of N_2O and may be a greater source per unit area than wetlands. Goodroad and Keeney (1984) found higher concentrations of N_2O in a drained marsh than in an undrained marsh, probably due to accelerated mineralization and decomposition in the former. Weller et al. (1994) found that while N_2O emissions from corn fields and riparian forests were equal in the fall, corn field emissions were three times higher than those of riparian forests in the spring.
4. It is likely that significant denitrification is currently occurring in the Gulf of Mexico hypoxic zone. Creation and restoration of anaerobic wetlands in the Mississippi River Basin would transfer some of this denitrification from the Gulf to the Mississippi River Basin. Production of nitrous oxide is ultimately related to the amount of nitrogen added to the basin by fertilizer use, soil mineralization, legumes, and other primary sources.
5. Denitrification of nitrous oxide is ultimately related to the amount of nitrogen added to the basin by fertilizer use, soil mineralization, legumes, and other primary sources.

CHAPTER 5

Research Needs

The Topic 5 team has identified the following research needs for reducing nutrient loads to the Gulf of Mexico:

1. More refined soil nitrogen testing procedures need to be developed and tested, including in-season testing of corn leaf tissue and the use of chlorophyll measurements, especially from remote-sensing platforms.
2. The newly developing concept of “precision farming” needs to be investigated thoroughly for its ability to reduce nitrate losses from an agricultural landscape. To date, little evidence exists to show that precision farming reduces nitrate losses to ground or surface water.
3. Alternatives to the traditional corn–soybean rotation and their effects on nitrate loss to subsurface drainage water should be investigated. This study should also include economic analyses.
4. The effects of variables, such as drainage tile spacing and depth, and the effectiveness of controlled drainage on nitrogen retention in poorly drained soils need to be determined through controlled experiments.
5. There is a critical need for better understanding of nitrogen behavior during floods and catastrophic events, particularly in ecotechnological methods for nitrate–nitrogen control, such as riparian zones and other wetlands.
6. Controlled large-scale experiments on the fate of nitrogen are needed on the reflooding of formerly tile-drained lands as restored wetlands.
7. Research is needed on the long-term efficacy of wetlands for denitrification, including the time required for organic carbon accumulation in created wetlands and the role that this accumulation has on denitrification.
8. There is a critical need for additional farm-scale studies on integration of crop land, riparian buffers, and wetlands to most effectively reduce nitrogen entry into streams. This information must be used by researchers in large-scale watershed models designed to determine the most effective placement of these management options throughout much larger watersheds to achieve multiple-objective goals.
9. There should be a comprehensive effort to determine the mix of different nitrate reduction strategies that gives the best nitrate reduction for the least cost. This effort should involve pilot studies in different parts of the Mississippi River Basin.
10. Additional study and modeling are needed to demonstrate the relationships among land subsidence, river diversion rates, and nitrogen uptake in the delta region of Louisiana.

11. A complete accounting of the production of the greenhouse gas N_2O both from increased wetland development as well as from drained and fertilized agricultural land is needed in comparative studies. A basin-wide study is also needed to compare these fluxes to the current production of N_2O in the rivers of the basin and in the hypoxic zone of the Gulf of Mexico.
12. Studies are needed on projected increases in population and other development in the Mississippi River Basin and on how these changes might offset reductions in nitrogen loadings from the basin.

CHAPTER 6

Conclusions and Recommendations

6.1 CONCLUSIONS

We have reviewed the most likely methods for reducing nitrogen loading to the Gulf of Mexico from the Mississippi River Basin. The scale of the watershed (at 40% of the conterminous United States) and the climatological and geologic heterogeneity of the basin make recommendations of specific methods and sites to implement them particularly difficult. Nevertheless, we conclude that a significant (> 50%) reduction of nitrogen loading to the Gulf is possible through the implementation of a number of proven techniques working in concert, including:

1. modification of farm practices to make the use of nitrogen from fertilizer, soil, and manure more effective and efficient;
2. the creation and restoration of wetlands and riparian ecosystems between farmland and streams and rivers, particularly in areas where concentrations of subsurface nitrate–nitrogen is highest;
3. the reflooding of former wetlands that are now contributing excessive loadings of nitrate–nitrogen due to their drainage;
4. the implementation of nitrogen controls on domestic wastewater treatment plants and on significant industrial sources;
5. flood control in the Upper Mississippi that involves retention of flood waters, rather than preventing flood waters from leaving the major river channel; and
6. the diversion of flood waters to backwaters of the Mississippi Delta and coastal wetlands.

6.2 RECOMMENDATIONS

Several of the following recommendations need to be implemented in concert if a major reduction in nitrogen loading to the Gulf of Mexico is expected. If policies are devised to implement only one or two of these recommended approaches, improvement in the hypoxia problem in the Gulf is not as likely.

1. A suite of on-farm practices for reducing discharges of nitrogen to streams and rivers should be implemented. These practices could lead to reductions of 15–20% in nitrogen sources to the Gulf. This would require approximately a 20% reduction in fertilizer nitrogen application through proper nitrogen crediting for legumes and manure, elimination of "extra" N to minimize risk, and use of realistic yield goals and proven yields when making nitrogen fertilizer recommendations. Other recommended management practices include optimum timing of fertilizer application; use of alternative crops, such as perennials; wider spacing of tile drains; and better management of livestock wastes, whether stored or applied to the land.

2. Reducing nonpoint sources of nitrogen from the Mississippi River Basin will also require extensive creation and restoration of riparian zones and/or wetlands. A major effort should be undertaken in the basin to restore or create 21,000 km^2 (5 million acres, or 0.7% of the basin) of wetlands and 78,000 km^2 (19 million acres, or 2.7% of the basin) of riparian forest, or some other combination of these two approaches, to achieve a combined 40% reduction of nitrogen loading in the Gulf.
3. The location and selection of type of riparian zone and wetland will be critical. Since much of the nonpoint-source nitrogen is currently entering surface waters through drainage tiles, wetlands should be strategically placed in watersheds to optimize nitrogen removal. For example, tile-drained farmlands that are prone to export high concentrations and fluxes of nitrate during high precipitation should be investigated for the possibility of drainage tile removal or interception to restore the natural hydrology to the land under various conservation easement and wildlife enhancement programs. The location of the wetlands should also try to optimize flood control and habitat provision.
4. Although point sources of nitrogen appear to be of little consequence (< 5%) in the Mississippi River Basin's overall nitrogen load, an effort to control these sources through tertiary treatment should become a formal policy for new wastewater treatment plants in the basin because nitrate concentrations are relatively high in treated wastewater and because these plants represent more easily controlled point sources. Furthermore, there may be opportunities for nitrogen trading whereby agricultural interests can "buy" credit from municipalities for these reductions in nitrogen loading to the basin.
5. The restoration of flood-prone lands in the Upper Mississippi River Basin to wetlands needs to be revisited and more seriously considered in light of the 1993 flood *and* the need to control nitrate–nitrogen to protect the Gulf. These wetlands would provide the triple advantages of retaining flood water, reducing nitrate–nitrogen loading to the Gulf, and providing needed wildlife habitat.
6. Nitrate reduction should become an important consideration in the design and operation of diversions of the Mississippi River for flood events in the Mississippi Delta in Louisiana. The State of Louisiana should consider the implications of the use of 0.4–1.0 million hectares (1.0–2.5 million acres) of inshore coastal areas (forested wetlands, marshes, and water bodies) for reducing nitrate in diverted waters. The most important benefit of such diversions would be to address the land loss problem, while a secondary benefit would be reducing nutrient discharge to the nearshore Gulf of Mexico.
7. Reductions of atmospheric nitrogen emissions beyond those now being implemented through the authority of the Clean Air Act Amendments are probably not warranted for controlling stationary and mobile sources of nitrogen, at least insofar as protection of the Gulf of Mexico is concerned.
8. There is a strong need for any nitrogen mitigation effort to be coupled to a comprehensive program of monitoring, research, and modeling. We need to know what practices work, and why, so that "adaptive management" of the hypoxia problem can be carried out.

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