Science for Solutions

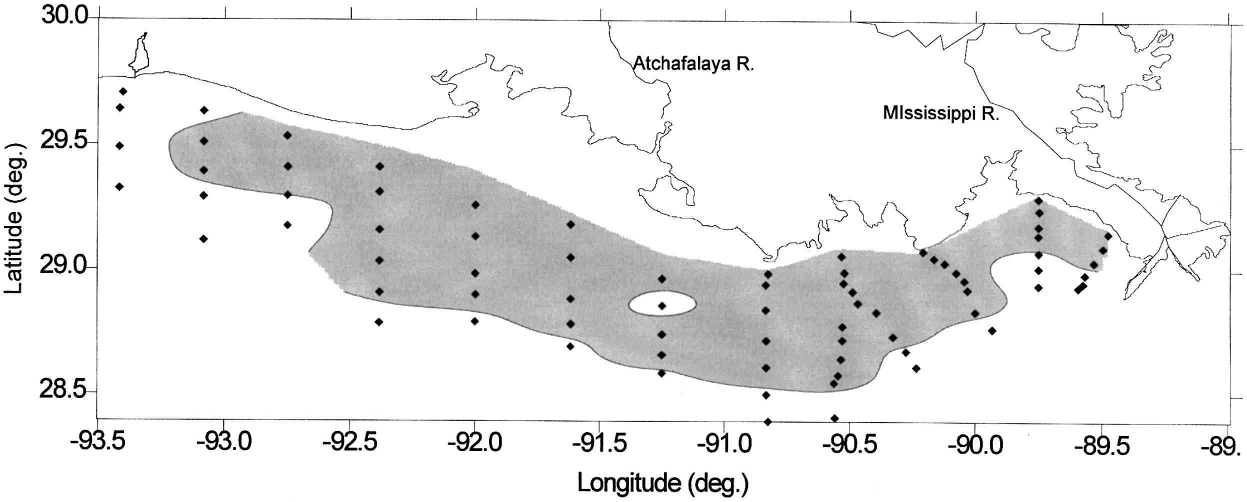
NOAA COASTAL OCEAN PROGRAM
Decision Analysis Series No. 15

Characterization of Hypoxia

Topic 1 Report for the Integrated Assessment on Hypoxia in the Gulf of Mexico

Nancy N. Rabalais, R. Eugene Turner, Dubravko Justić,
Quay Dortch, and William J. Wiseman, Jr.

May 1999



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

GULF OF MEXICO HYPOXIA ASSESSMENT

This report is the first 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: Extent of bottom water with dissolved oxygen less than 2 mg/l for July 1996 (based on data of Rabalais, Turner, and Wiseman).

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Acknowledgments

Over the years, numerous agencies and institutions have supported the research that provided the bases for this synthesis: the National Oceanic and Atmospheric Administration's (NOAA's) Coastal Ocean Program, Ocean Assessments Division, and National Undersea Research Center; the U.S. Department of the Interior's Minerals Management Service; the Louisiana Board of Regents' Education Quality Support Fund; the Louisiana Sea Grant College Program; the U.S. Department of Energy's Climate Change Program; the National Science Foundation; the U.S. Environmental Protection Agency's Gulf of Mexico Program; Louisiana State University; and Louisiana Universities Marine Consortium (LUMCON). Over the years we have also had the help of many research assistants, research associates, postdoctoral associates, visiting scientists, volunteers, divers, data managers, ship captains, and ship's crew.

Special thanks are due to Ben Cole of LUMCON; without his management of the data and help with graphics, this report could not have been completed. Don Scavia, Mike Dowgiallo, and John Field of NOAA provided the guidance for the initiation, development, and completion of the report. Fred Bryan and Don Boesch provided the editorial oversight for seven anonymous reviewers. Gene Meier of the Gulf of Mexico Program provided oversight for the program that managed the funds from the Inter-Agency Hypoxia Task Force that supported this analysis.

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

Nutrient overenrichment from human activities is one of the major stresses affecting coastal ecosystems. There is increasing concern in many areas around the world that an oversupply of nutrients from multiple sources is having pervasive ecological effects on shallow coastal and estuarine areas. These effects include reduced light penetration, loss of aquatic habitat, harmful algal blooms, a decrease in dissolved oxygen (or hypoxia), and impacts on living resources. The largest zone of oxygen-depleted coastal waters in the United States, and the entire western Atlantic Ocean, is found in the northern Gulf of Mexico on the Louisiana–Texas continental shelf. This zone is influenced by the freshwater discharge and nutrient flux of the Mississippi River system.

This report describes the seasonal, interannual, and long-term variability in hypoxia in the northern Gulf of Mexico and its relationship to nutrient loading. It also documents the relative roles of natural and human-induced factors in determining the size and duration of the hypoxic zone.

GENERAL DIMENSIONS

Hypoxia covers broad regions of the shelf for extended periods in mid-summer. In 1985–92, the mid-summer bottom areal extent of hypoxic waters ($\leq$ 2 mg/l O_2, or ppm) averaged 8,000–9,000 km^2; in 1993–97 it increased to 16,000–18,000 km^2. The estimated extent was 12,500 km^2 in mid-summer of 1998.

A compilation of 13 mid-summer shelf-wide surveys (1985–97) demonstrates that the frequency of occurrence of hypoxia is higher to the west of the discharges of the Mississippi and Atchafalaya Rivers in a down-current direction from their influence. The areal extent of mid-summer hypoxia is a minimal estimate and provides little information on the persistence of the zone over large areas, or the temporal sequence of physical and biological processes that preceded the mapping. Surveys along a transect off Terrebonne Bay on the southeastern Louisiana shelf on a monthly basis from 1985 to 1998 provide better temporal resolution and a time series suitable for examining monthly- and seasonal-scale differences and relationships with variability in Mississippi River system discharge and nutrient flux.

Hypoxic waters are most prevalent from late spring through late summer, and hypoxia is more widespread and persistent in some years than in others. Hypoxic waters are distributed from shallow depths near shore (4 to 5 m) to as deep as 60 m, but are present more typically between 5 and 30 m. Hypoxia occurs mostly in the lower water column but encompasses as much as the lower half to two-thirds of the column.

Continuous time-series data for the bottom waters in the core of the hypoxia region show (1) the gradual decline in oxygen in the spring with interruptions due to wind-mixing events, (2) persistent hypoxia and often anoxia for extended parts of the record from May through September, (3) occasional summer upwelling of oxygenated water from the outer shelf, and (4) the seasonal disruption of low oxygen in the fall from tropical storms or cold fronts.

RIVER DISCHARGE AND FLUX OF MATERIALS

The Mississippi River system encompasses 41% of the conterminous United States and delivers an average of 580 km^3 of fresh water to the Gulf of Mexico yearly, along with sediment yields of 210 x 10^6 t/yr, 1.6

x 10^6 t/yr nitrate (N), 0.1 x 10^6 t/yr phosphorus (P), and 2.1 x 10^6 t/yr silica (Si). The best current knowledge is that the outflows of the Mississippi and Atchafalaya Rivers dominate the nutrient loads to the continental shelf where hypoxia is likely to develop.

The 1820–1992 average discharge rate (decadal time scale) for the Mississippi River at Vicksburg is remarkably stable near 14,000 m^3/s, despite significant interannual variability and some decadal trends. Since the 1700s humans have altered the morphology and flow of the Mississippi River so that now 30% is diverted to the Atchafalaya that also captures the flow of the Red River. The discharge of the Atchafalaya increased during the period 1900–1992, primarily as a result of the tendency for the Atchafalaya to capture more of the flow of the Mississippi (until stabilized at 30% in 1977). An effect likely to occur in the offshore region as a result of increased flow through the Atchafalaya Delta is an increase in stratification west of Atchafalaya Bay and further westward into Texas. Managing the birdfoot delta for greater discharge through Southwest Pass may have increased inputs to the Mississippi River Bight and offset siphoning of discharge to the Atchafalaya. A slight increase in Mississippi River discharge for 1900–1992 is accounted for by an increased discharge in September through December, a period that is much less important in the coastal ocean than spring and summer in the timing of important biological processes that lead to the development of hypoxia or the physical processes important to its maintenance.

Mississippi River nutrient concentrations and loading to the adjacent continental shelf have changed dramatically this century, with an acceleration of these changes in the last four decades, depending on the constituent of concern. The mean annual concentration of nitrate was approximately the same in 1905–6 and 1933–34 as in the 1950s, but it has doubled (or tripled, depending on the comparative periods) from the 1950s to 1960s. The increase in total nitrogen is almost entirely due to changes in nitrate concentration. The mean annual concentration of silicate was approximately the same in 1905–6 as in the early 1950s, then it declined by 30–50%, depending on the analysis or period of record. Concentrations of nitrate and silicate appear to have stabilized, but trends are masked by increased variability in the 1980s and 1990s data. There are no substantial records of total phosphorus concentrations in the lower Mississippi River before 1973, and subsequent values vary greatly among years. However, application of a linear least-squares regression on the 1973–87 data indicates a twofold increase in the total phosphorus concentration. While there was no pronounced seasonal peak in nitrate concentration before 1960, there was a spring peak from 1975 to 1985. Before the 1960s, nitrogen flux closely paralleled river discharge—a pattern that still holds, but the load of nitrogen per volume discharge is greater than historically. There is no doubt that the concentration and flux of nitrogen (per unit volume discharge) have increased from the 1950s to 1960s, especially in the spring.

The proportions of dissolved Si, N, and P in the lower Mississippi River have changed historically such that they now closely approximate the Redfield ratio (Si:N:P = 16:16:1). Thus any single nutrient is more likely to be limiting to phytoplankton production now than historically. Another reasonable hypothesis that follows a more balanced nutrient composition is that surface offshore primary productivity has increased. Fluctuations in the Si:N ratio within the riverine effluents and the offshore waters can affect diatom production and are believed to be major determinants in the coastal food web structure on a seasonal basis, with major implications for oxygen and carbon cycling. Thus, long-term effects in the offshore ecosystem are likely a result of changes in constituents, primarily nitrogen as nitrate, and not the amount of freshwater discharge or alterations in freshwater delivery.

INTERACTION OF PHYSICS AND BIOLOGY

The physics of the coastal ecosystem and its biological processes are linked and related to the freshwater discharge and nutrient flux of the Mississippi River system. The physics of the nearshore Gulf of Mexico define where hypoxia can occur, and the biological processes of carbon production, flux, and respiration that lead to oxygen depletion. The high freshwater discharge, general circulation patterns of the Louisiana shelf, and the presence of the Louisiana Coastal Current dictate a stratified system for much of the year, interrupted on occasion by wind-mixing events, notably tropical storms and winter cold fronts.

NUTRIENT-ENHANCED PRODUCTIVITY

The evidence for nutrient-enhanced primary production in the northern Gulf of Mexico and its linkage with oxygen depletion in the lower water column comes from information on a variety of scales—experiments for a parcel of water from a particular locale over a limited time to more integrative measures of ecosystem response (e.g., net production, carbon flux and respiration) and change over broader spatial and temporal scales. The concentrations, total loads, and ratios of nutrients (nitrogen, phosphorus, and silica) delivered to the coastal ocean influence the productivity of the phytoplankton community as well as the types of phytoplankton that are most likely to grow.

The nutrient most relevant to overall phytoplankton production over the broad region fueling hypoxia is nitrogen, and nitrate–nitrogen makes up approximately two-thirds of the total nitrogen input from the Mississippi River. Silica and phosphorus may also be limiting at some times and places. There is clear evidence that primary production in shelf waters near the delta and some distance from it are significantly correlated with nutrient inputs (nitrate + nitrite and orthophosphate). Similar relationships exist with net production (an indicator of the amount of carbon available for export to the lower water column and sediments) and nitrate flux. There is also a strong relationship between the net production in surface waters, the amount of carbon exported, the accumulation rates of carbon, and the depletion of oxygen in bottom waters. Spatial and temporal variability in these components is closely related to the amplitude and phasing of Mississippi River discharge and nutrient fluxes. Thus, there are clear lines of evidence for nitrogen- (particularly nitrate-) driven phytoplankton production that leads to hypoxia. Although the Mississippi and Atchafalaya Rivers discharge organic matter to the shelf, the principal source of carbon reaching the bottom waters in the northern Gulf influenced by the river effluent and characterized by hypoxia is from *in situ* phytoplankton production.

LONG-TERM CHANGES IN THE COASTAL ECOSYSTEM

It follows, and is supported with evidence from long-term data sets and the sedimentary record, that increases in riverine dissolved inorganic nitrogen concentrations and loads are highly correlated with indicators of increased productivity in the overlying water column—i.e., eutrophication of the continental shelf waters, and subsequent worsening of oxygen stress in the bottom waters. Evidence comes from changes in diatom production, increased accumulation of diatom remains in the sediments, increased carbon accumulation in the sediments, decreased diversity of selected benthic fauna, and relative changes in selected benthic fauna that indicate a worsening oxygen environment.

Human activities in the watershed undoubtedly changed the natural functioning of the Mississippi River system. Century-long patterns of freshwater discharge are not evident; thus, the long-term changes on the Louisiana shelf are linked to the quality of the discharge (nutrient loads and ratios of nutrients) and not the amount. Century-long changes are evident in some of the retrospective analyses; however, the most dramatic and accelerating changes have been since the 1950s, when nitrogen loads began to increase, primarily from nitrate inputs, and eventually doubled to tripled over their historic values. The fact that the most dramatic changes in the continental shelf ecosystem have occurred since the 1950s and are coincident with an increase in nitrate load points to that aspect of human ecology for future management scenarios.

Evidence associates increased coastal ocean productivity and worsening oxygen depletion with changes in landscape use and nutrient management that resulted in nutrient enrichment of receiving waters. Nutrient flux to coastal systems, while essential to the overall productivity of those systems, has increased over time due to anthropogenic activities and has led to broad-scale degradation of the marine environment.

NUTRIENT INTERVENTION

The northern Gulf of Mexico adjacent to the discharge of the Mississippi River system is an example of a coastal ocean that has undergone eutrophication (an increase in the rate of primary production) as a result of increasing nutrients and that has worsened hypoxic conditions on century-long and accelerating recent decadal time scales. Models that link Mississippi River discharge with Gulf of Mexico hypoxia demonstrate worsening hypoxia in bottom waters with increased freshwater discharge, and even worse hypoxia with additional nitrogen accompanying the increased discharge. Conversely, the models show that a reduction in oxygen demand in the lower water column will result from a reduction in the nitrogen (and to a lesser degree the phosphorus) load to the surface waters. In other words, hypoxia in the northern Gulf of Mexico can be alleviated to some degree by a reduction in the nutrient loading.

Whole-system management of the entire watershed where most of the changes have occurred over the last several decades is a necessary step in alleviating the problems in the Gulf of Mexico. The ability to detect changes in the coastal system (given any nutrient reductions) will be complicated by an inherently variable biological system and extreme events. In addition, the eutrophic state may be persistent and recovery may be slow. Still, there are several success stories for improvement of estuarine and coastal ecosystems in response to nutrient abatement in the watershed or in direct discharges to the system, and similar activities on larger coastal systems with a much larger watershed, while daunting, are worthwhile and achievable.

CHAPTER 1

Introduction

Nutrient overenrichment from human activities is one of the major stresses affecting coastal ecosystems. There is increasing concern in many areas around the world that an oversupply of nutrients from multiple sources is having pervasive ecological effects on shallow coastal and estuarine areas. These effects include reduced light penetration, loss of aquatic habitat, harmful algal blooms, a decrease in dissolved oxygen (or hypoxia), and impacts on living resources.

Prolonged oxygen depletion can disrupt benthic and demersal communities and cause mass mortalities of aquatic life (Diaz and Rosenberg 1995). Hypoxia affects living resources, biological diversity, and the capacity of aquatic systems to support biological populations. When oxygen levels fall below critical values, organisms capable of swimming (e.g., demersal fish, portunid crabs, and shrimp) usually evacuate the area. The stress on less mobile fauna varies, but they also experience stress or die as oxygen concentrations fall to zero. Important fishery resources are variably affected by direct mortality, forced migration, reduction in suitable habitat, increased susceptibility to predation, changes in food resources, and disruption of life cycles (Diaz and Solow 1999).

Hypoxia occurs in many parts of the world's aquatic environments. Hypoxic and anoxic (oxygen-deprived) waters have existed throughout geologic time, but their occurrence in shallow coastal and estuarine areas appears to be increasing, most likely accelerated by human activities. The largest zone of oxygen-depleted coastal waters in the United States, and the entire western Atlantic Ocean, is in the northern Gulf of Mexico on the Louisiana–Texas continental shelf. From 1993 to 1997, the size of the Gulf of Mexico hypoxic zone was consistently greater than 16,000 km^2 in mid-summer, but covered 12,480 km^2 in mid-summer of 1998 (Rabalais et al. 1998, unpublished data). The zone ranks third in area behind the northwestern shelf of the Black Sea and the Baltic basins (Boesch and Rabalais 1991). The hypoxic zone in the northern Gulf of Mexico (average for 1993–97) is about the size of the state of New Jersey or the states of Rhode Island and Connecticut combined. Its total extent on the bottom is twice the total surface area of the whole Chesapeake Bay, and its volume is several orders of magnitude greater than the hypoxic water mass of Chesapeake Bay (Rabalais 1998a). For a comparison with a portion of geography from the middle of the United States, the distance across the hypoxic zone from the Mississippi River onto the upper Texas coast is equal to the distance between Des Moines, Iowa, and Chicago, Illinois.

The watershed that drains through the Mississippi and Atchafalaya Rivers is also immense. The Mississippi River system ranks among the world's top 10 rivers in length, freshwater discharge, and sediment delivery and drains 41% of the lower 48 United States (Milliman and Meade 1983; Meade 1995). Thus, the dimensions of the problem and the drainage system that affect it are of much greater magnitude than most nutrient-driven eutrophication problems elsewhere. Model simulations, research studies, empirical relationships, and retrospective analyses of the sedimentary record have produced considerable evidence that nutrient loading from the Mississippi River system is the dominant factor in controlling the extent and degree of hypoxia.

This report describes the seasonal, interannual, and long-term variations in hypoxia in the northern Gulf of Mexico, and its relationship to nutrient loading. It also documents the relative roles of natural and human-induced factors in determining the size and duration of the hypoxic zone.

CHAPTER 2

Methods

This synthesis was generated from a review of the scientific literature, reports, and compilation of numerous data sets. Primary data are from the hypoxia studies of Rabalais, Turner, and Wiseman beginning in 1985 and the phytoplankton community studies of Dortch beginning in 1990. The National Oceanic and Atmospheric Administration's (NOAA's) Nutrient Enhanced Coastal Ocean Productivity (NECOP) program generated a tremendous amount of data on Mississippi River/Gulf of Mexico interactions (Atwood et al. 1994). Another study with significant hydrographic and biological data was the Louisiana–Texas Physical Oceanography Program (LATEX) (Murray 1998). Details of data collection and methodology are provided in the cited papers and reports or accompany the various data sets as submitted to NECOP data management (Hendee 1994) or to NOAA's National Oceanographic Data Center for both NECOP and LATEX data. Additional analyses of existing data were not conducted for this synthesis, but several needed data syntheses are identified throughout the text.

Hypoxia is operationally defined as dissolved oxygen levels below 2 mg/l, or ppm, for the northern Gulf of Mexico. This is the level below which trawlers usually do not capture any shrimp or demersal fish (Leming and Stuntz 1984; Pavela et al. 1983; Renaud 1986). When dissolved oxygen values are below 2 mg/l, they are often less than 1 mg/l—a severe level that is stressful or lethal to benthic macroinfauna. Dissolved oxygen of 2 mg/l equates to 1.4 ml/l, and approximates 20% oxygen saturation in northern Gulf of Mexico waters. Oxygen-deficient (less than 100% oxygen saturation) waters are more widespread than indicated by the 2 mg/l cutoff, but for consistency a value of 2 mg/l is used throughout this synthesis.

Surveys of the mid-summer extent of hypoxia over a standard station grid (Figure 2.1) provide a broad-scale view of the extent of the hypoxic zone for interannual comparisons. A single cruise of this nature in mid-summer, however, provides little information on the persistence of the zone over large areas, or the temporal sequence of physical and biological processes that preceded the cruise. Surveys along transect C on the southeastern shelf off Terrebonne Bay from nearshore to 30-m water depth on a monthly or twice monthly basis from 1985 to 1998 provide better temporal resolution over a more limited area of the shelf. Transect C data provide a time series suitable for examining monthly- and seasonal-scale differences over a representative area of the Louisiana shelf and relationships with variability in Mississippi River discharge and nutrient flux. Station C6* (includes data from C6A, C6B, and C6) in the core of the hypoxic zone on transect C in 20-m water depth is the site of an instrument mooring and more detailed data collection and experiments. The temporal resolution for selected parameters is the best for station C6* and can be extrapolated for a portion of the Louisiana shelf, but not the entire shelf. Logistical or funding constraints and/or technology improvements altered data acquisition through the years, but the data are consistent, precise, and accurate.

Standard data collections included hydrographic profiles for temperature, salinity, dissolved oxygen, and optical properties. Water samples for chlorophyll *a* and phaeopigments, nutrients, salinity, suspended sediment, and phytoplankton community composition were collected from the surface, near-bottom, and variable middle depths.

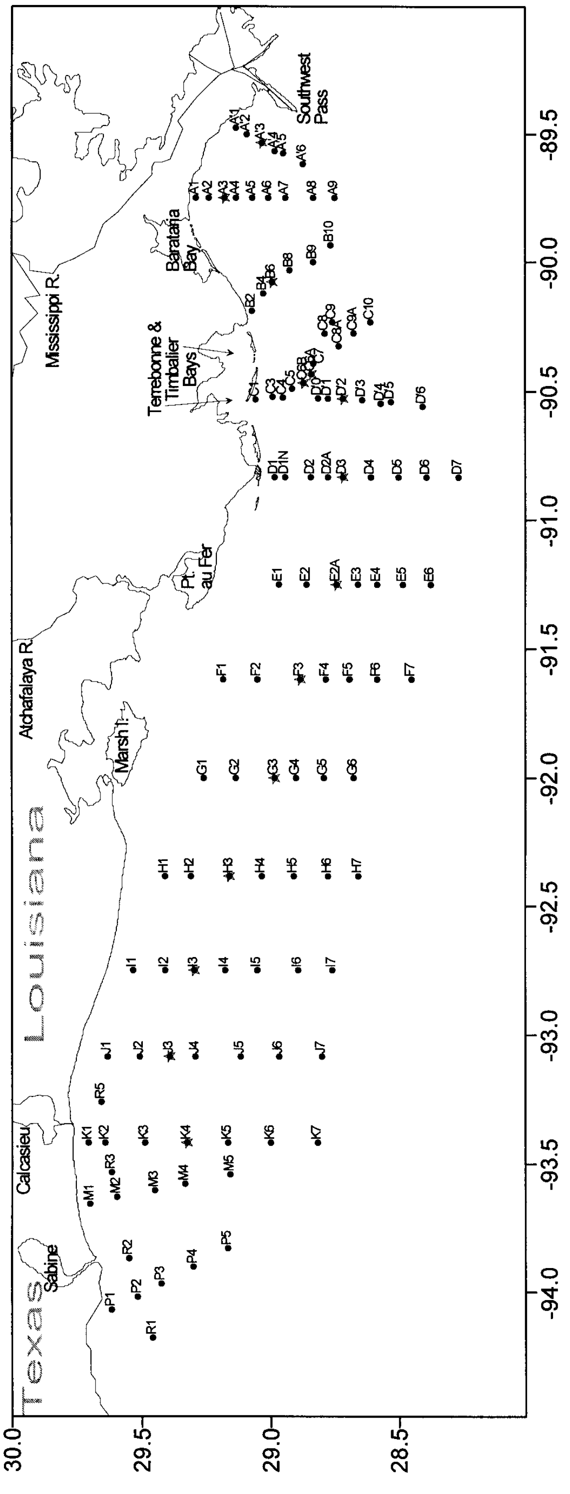


FIGURE 2.1. Distribution of stations for mid-summer shelf-wide surveys and more frequent sampling along transect C. NOTE: Stations C6A and C6B are locations of moored instruments; station C6, not shown, is between C6A and C6B. Composite data for stations C6A, C6B, and C6C comprise values for station C6*. Starred stations represent those that approximate the 20-m isobath.

CHAPTER 3

Dimensions and Variability of Hypoxia

Hypoxic waters are distributed from shallow depths near shore (4–5 m) to as deep as 60 m, but are present more typically between 5 and 30 m. The distance offshore that bounds hypoxic water masses is contoured by the slope of the continental shelf. On the southeastern Louisiana shelf, where the shelf slopes more steeply toward the Mississippi Canyon, hypoxia extends only 55 km from shore. On the central and southwestern Louisiana shelf, where the continental shelf is broader and the depth gradient is more gradual, hypoxic bottom waters may extend as far as 130 km offshore.

Hypoxia occurs not only at the bottom near the sediments, but well up into the water column (see Sections 3.2 and 3.4). Depending on the depth of the water and the location of the pycnocline(s), hypoxia may encompass from 10% to over 80% of the total water column, but normally encompasses 20–50%. Hypoxia may reach to within 2 m of the surface in a 10-m water column, or to within 6 m of the surface in a 20-m water column. Anoxic bottom waters can occur, along with the release of toxic hydrogen sulfide from the sediments.

3.1 MID-SUMMER EXTENT

Since 1985 the distribution of hypoxia on the Louisiana shelf has been mapped during mid-summer cruises (usually mid-July to early August, during the expected maximal extent of hypoxia). (Appendix A presents contours and hypoxic areas for each year.) The sampling grid is similar from year to year (abbreviated cruises were conducted in 1988 and 1989) and covered within a five-day period. On any cross-shelf transect, sampling is conducted from about 5-m water depth, or as nearshore as the research vessel can safely navigate, to as far offshore as necessary to delineate the distribution of hypoxia.

For the period 1985–92, the zone of hypoxia was usually in a configuration of disjunct areas situated to the west of the deltas of the Mississippi and Atchafalaya Rivers, and the bottom area averaged 7,000–9,000 km^2. The area of hypoxia in mid-summer 1988 was confined to a single station off Terrebonne Bay. A reduced area was mapped in 1989 as part of a NURC (National Undersea Research Center) cruise. Bottom-water hypoxia was continuous across the Louisiana shelf in mid-summer of 1993–97, and the bottom area (16,000–18,000 km^2) was twice as large as the 1985–92 average. The estimated bottom area mapped in mid-summer 1998 was comparable to the size in 1991. The 1998 hypoxia was concentrated on the eastern and central Louisiana coast from the Mississippi River Delta to Marsh Island near Atchafalaya Bay and in deeper water than usual (up to 50+ m deep).

A compilation of 13 mid-summer shelf-wide surveys (1985–97) (Figure 3.1) demonstrates that the frequency of occurrence of hypoxia is higher to the west of the discharges of the Mississippi and Atchafalaya Rivers in a down-current direction from the influence of the freshwater discharge and nutrient flux. Other gradients in biological parameters and processes are also evident in a decreasing gradient away from the river discharges (see Chapter 6).

Hypoxia on the upper Texas coast is usually an extension of the larger hypoxic zone off Louisiana (maps in Section 3.2), although isolated areas may be found farther to the south (e.g., off Freeport and Matagorda Island areas) (Harper et al. 1981, 1991). Isolated conditions may be an artifact of the sampling, and very few systematic surveys have been conducted in this area, with the exception of the summer SEAMAP cruises (Gulf States Fisheries Commission 1982 et seq.). Mid-summer SEAMAP cruises documented hypoxia on the Texas coast in small, isolated areas in 1983, none in 1984–85, and in most years during 1991–97 (K. Craig, unpublished data). Most instances of hypoxia along the Texas coast are infrequent, short-lived, and limited in extent. There are no records of hypoxia south of the Matagorda Island, Texas, area (Rabalais 1992, updated with unpublished SEAMAP data of K. Craig personal communication).

Hypoxia has been documented off Mississippi Sound during high stages of the Mississippi River and off Mobile Bay in bathymetric low areas (Rabalais 1992). There are usually more reports in flood years or when more Mississippi River water moves to the east of the birdfoot delta. This was the case in 1993 off Mobile Bay and Mississippi Sound (M. Van Hoose, personal communication; Waller 1998). Of the 2,659 trawl, plankton, and videotaped trap stations occupied in the north-central Gulf of Mexico from the Mississippi River to Mobile Bay during SEAMAP cruises from 1984 to 1996 (Gulf States Marine Fisheries Commission 1982 et seq.), only 32 (or 0.012%) had bottom oxygen readings less than 2 mg/l (Waller 1998). While similar physical and biological processes are important in the development and maintenance of hypoxia on both sides of the Mississippi River Delta, the occurrence is much more widespread, persistent, and severe to the west of the delta. From limited data where both sides of the delta were surveyed for hydrographic conditions, including dissolved oxygen (Ragan et al. 1978; Turner and Allen 1982a), there is no evidence that the area of low oxygen forms a continuous band around the delta.

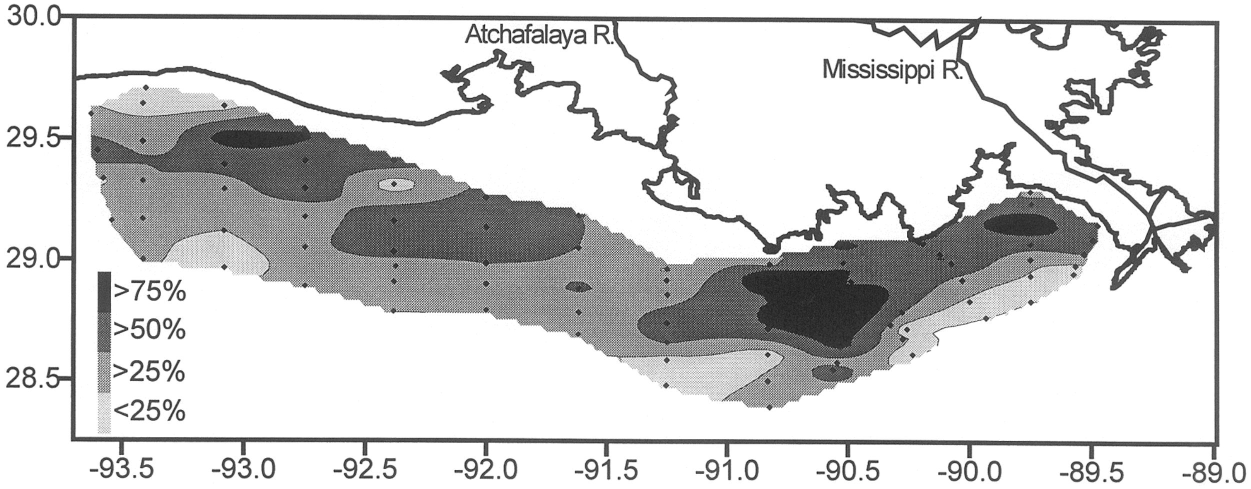


FIGURE 3.1. Distribution of frequency of occurrence of mid-summer hypoxia over the 60- to 80-station grid from 1985 to 1997. (*Data from Rabalais, Turner, and Wiseman hypoxia monitoring cruises.*)

A comparison of annual estimated areal extent (Figure 3.2) was generated from data collected along the standard 60- to 80-station grid of the Rabalais, Turner, and Wiseman hypoxia studies. (See Appendix A for station locations for each year.) The entire grid was seldom fully sampled, and additional stations were occasionally added to the grid. Time or other logistical constraints often prevented the complete mapping of the extent of hypoxia, either in the offshore direction or to the west. Thus, the areal extent of hypoxia generated from these surveys was a minimal estimate. The area estimations varied within a summer, and they should not be overinterpreted in making year-to-year comparisons or for identifying trends (Section 3.7).

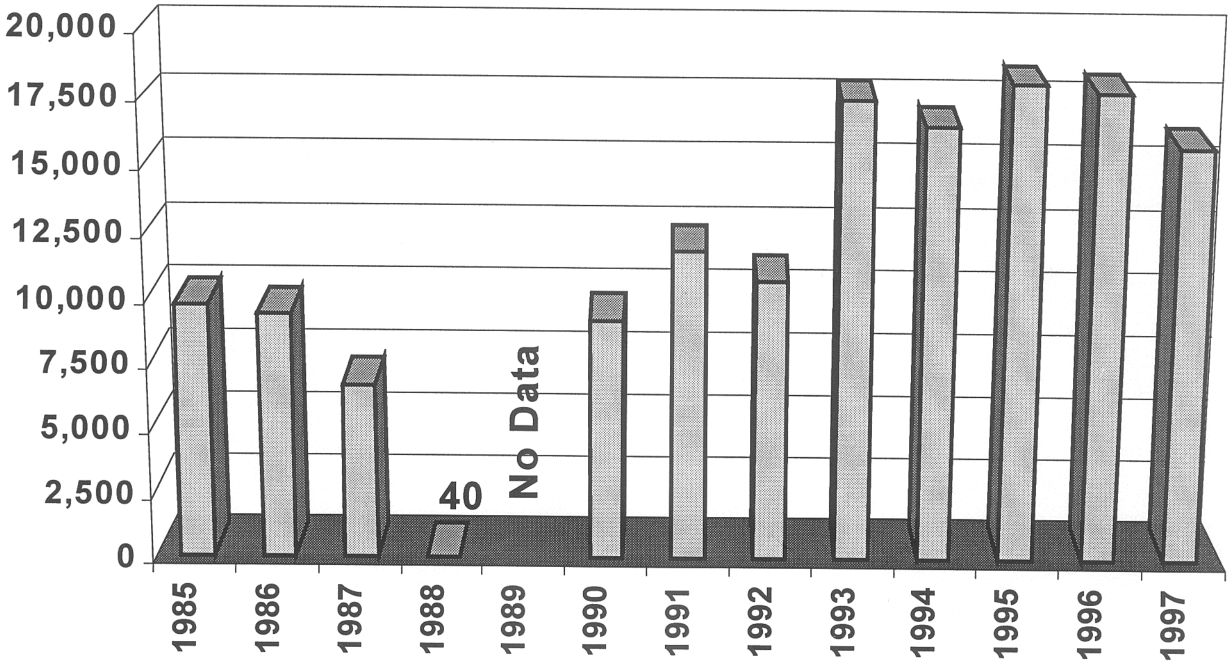


FIGURE 3.2. Histogram of estimated areal extent of bottom-water hypoxia (< 2 mg/l) for mid-summer cruises in 1985–97. (*Modified from Rabalais et al. 1998.)*

3.2 VARIABILITY IN MID-SUMMER

While single mid-summer surveys provide a broad-scale description of the extent of hypoxia, they do not provide information on the temporal persistence of the zone over a large area. Multiple mid-summer surveys of hypoxia were conducted in 1993 and 1994 with funding from NECOP and the LATEX Mississippi River plume hydrography study (Rabalais 1998b). Although the cruise tracks differed, the data provide evidence of the persistence of hypoxia on a broad scale, with some changes in intensity or distribution related to specific hydrographic processes. The same general area from the Mississippi River Delta to the upper Texas coast was surveyed on the NECOP cruises. LATEX station grids were chosen to define the coastal plumes of the Mississippi and Atchafalaya Rivers. Closure of oxygen isopleths at 2 mg/l was not feasible during the LATEX cruises, but was mostly accomplished for the NECOP cruises. Portions of transect C (NECOP) and line S1 (LATEX) and other cross-shelf transects overlapped and were suitable for comparisons. A third cruise was conducted during the NECOP study in July 1993 (Bratkovich et al. unpublished data) (Figure 3.3) and provided one more example of the persistence of hypoxia over large areas (at least in 1993) with the configuration of the areas changing over the course of a month.

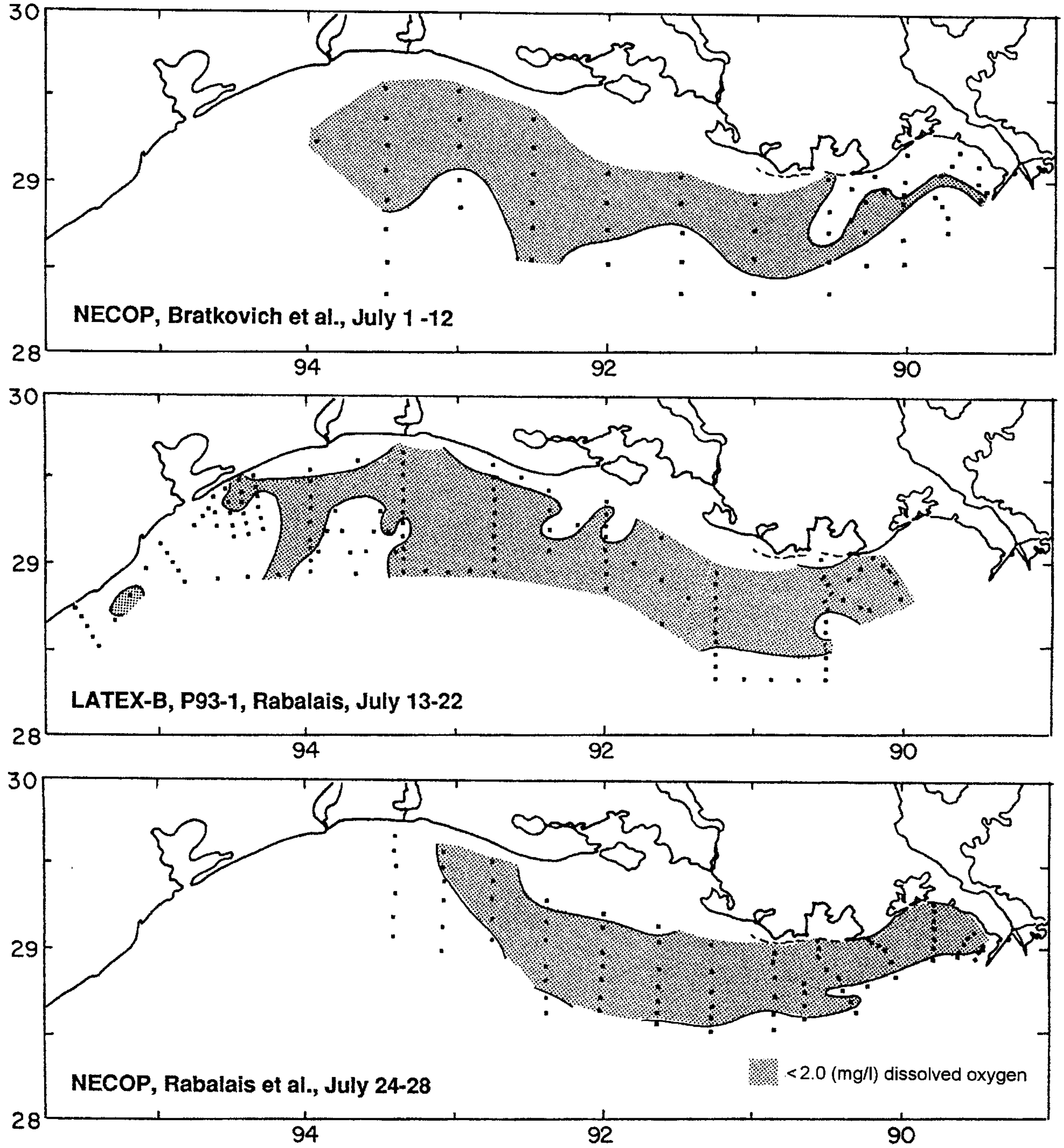


FIGURE 3.3. Comparison of bottom-water oxygen less than 2 mg/l for three cruises on the Louisiana–Texas shelf in July 1993. (*Top panel, Bratkovich et al. unpublished data; middle panel, Rabalais 1998b; lower panel, Rabalais et al. 1998.*)

There were extensive areas of hypoxia during multiple mid-July cruises in 1993 and 1994. During the 13–21 July 1993 cruise, the area of hypoxia extended from the transects off Terrebonne and Timbalier Bays on the east to the western end of the study area off Freeport, Texas (Figure 3.4). During the 24–28 July 1993 cruise, the hypoxic water mass did not extend as far to the west as during the previous cruise. Persistent winds from a southerly and southwesterly direction displaced much of the surface waters to the east (along with lower salinity and higher nutrients) as well as bottom water and the bottom-water hypoxia (Rabalais et al. 1998; Murray 1998). Although hypoxia was extensive during the 13–21 July cruise, it was located near the seabed and was somewhat patchy in distribution. One week later, hypoxia had become well developed, was present in much of the lower water column, was severe (well below 0.5 mg/l) at many stations, and was a continuous water mass. Data from stations along the C, D´, and D transects reoccupied during the first week of August 1993 (Rabalais et al. unpublished data) revealed: (1) a more extensive area of bottom-water hypoxia, (2) an increase in the frequency of dissolved oxygen < 0.5 mg/l and (3) a greater prevalence of anoxic conditions—i.e., detection of hydrogen sulfide in bottom-water samples.

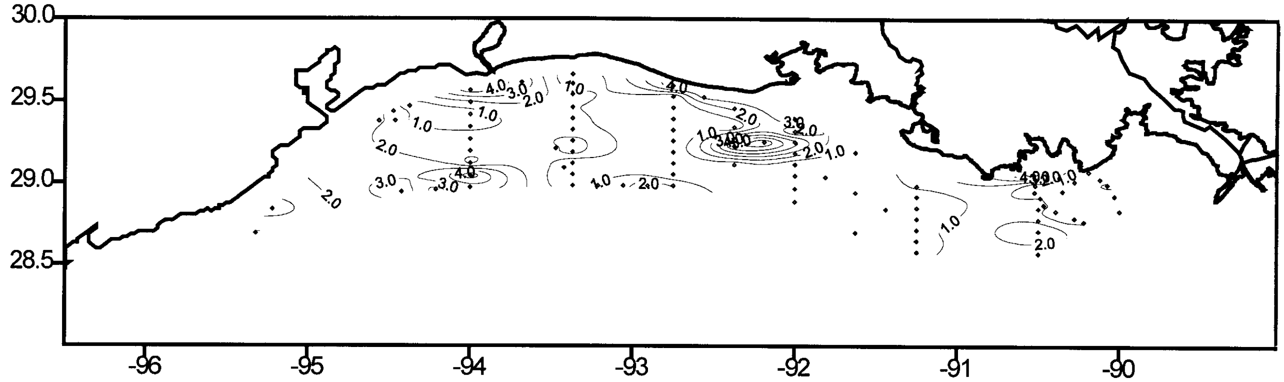
Equally extensive and severely depleted oxygen zones were documented in mid- to late July 1994 (Figures 3.5 and 3.6). Hypoxic bottom waters extended farther to the west off the Calcasieu and Sabine estuaries during the 12–18 July cruise and also at stations off Galveston. The 24–29 July cruise documented low oxygen conditions as far west as the Calcasieu estuary in 1994 that were very similar to the late July 1993 distribution. Hypoxia was displaced somewhat farther offshore during late July 1994 compared to the previous week. Cross-shelf transect comparisons between the 1994 LATEX and NECOP cruises revealed fairly consistent distributions of hypoxic waters and water-column structure through time.

Differences in 1993 from 13–21 July LATEX and 24–28 July NECOP cruises were evident as (1) increasingly more depleted oxygen concentrations, (2) a hypoxic zone encompassing more of the lower water column, (3) an eastward shift of the lower salinity surface waters from Atchafalaya Bay to off Terrebonne Bay, and (4) a shift eastward of the bottom-water hypoxia on the southwestern Louisiana shelf. Strong winds and high waves were present in the study area in mid-June because of the passage of a tropical storm across the Bay of Campeche. This wind-mixing event resulted in a reaeration of the water column that persisted into early July. During the progression of the LATEX and NECOP cruises, high flow of the Mississippi River persisted, winds calmed, stratification became re-established, and oxygen-consuming processes in the lower water column led to progressively lower oxygen concentrations in more of the water column. Figures 3.7–3.11 provide specific examples illustrating the above scenarios.

3.3 SHELF-WIDE DISTRIBUTION IN SPRING AND FALL

In addition to shelf-wide surveys during the maximal development of hypoxia, similar cruises (both LATEX and NECOP) were conducted in April, September, and October of selected years. Results from these surveys provide information on hypoxia over a large geographic region during the periods when it is likely to be developing or dissipating.

LATEX July 13-21, 1993
Bottom Dissolved Oxygen (mg/L)



NECOP July 24-30, 1993
Bottom Dissolved Oxygen (mg/L)

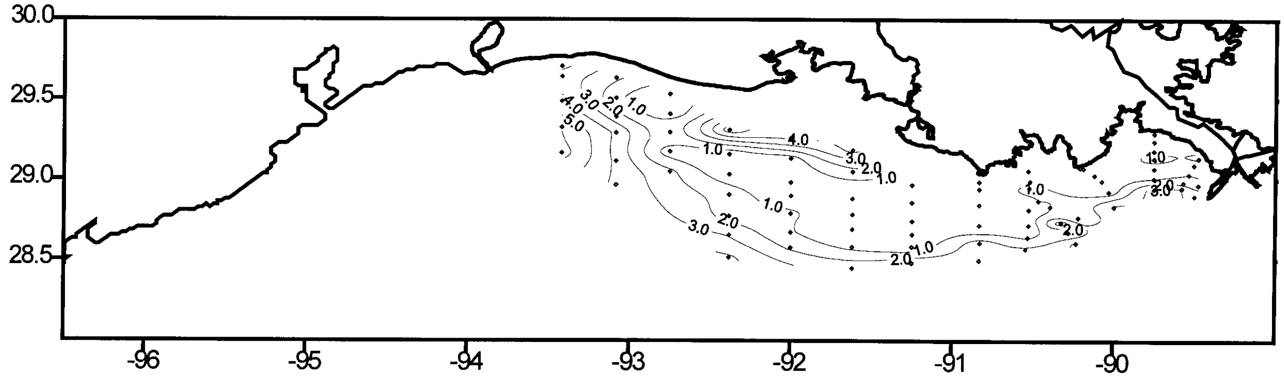
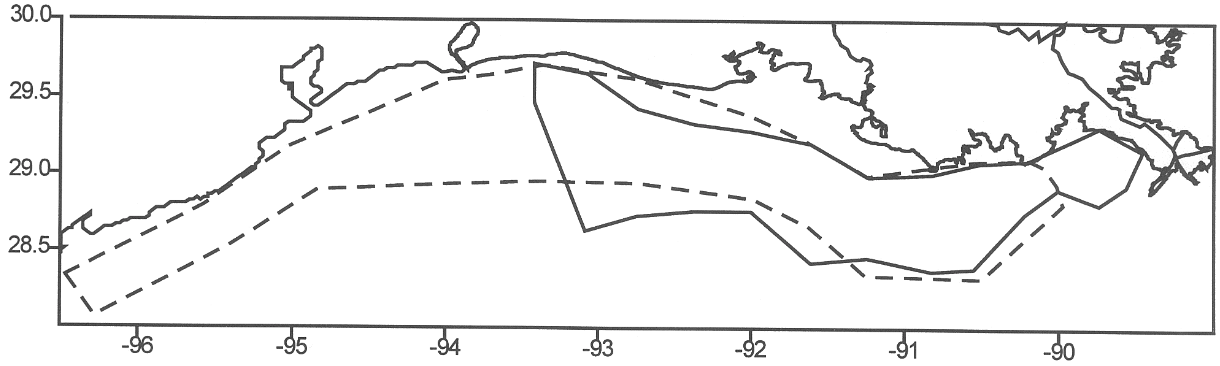
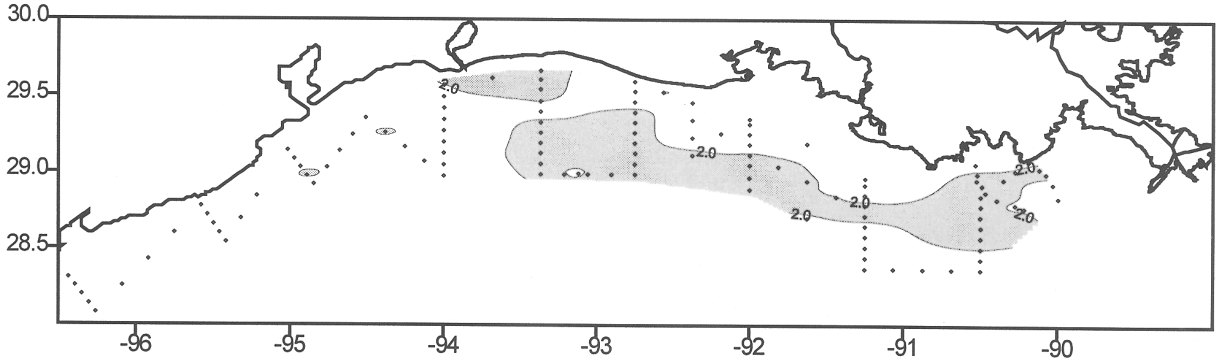


FIGURE 3.4. Bottom-water dissolved oxygen for July 1993 LATEX and NECOP cruises. (*Modified from Rabalais 1998b.)*



LATEX July 1994



NECOP July 1994

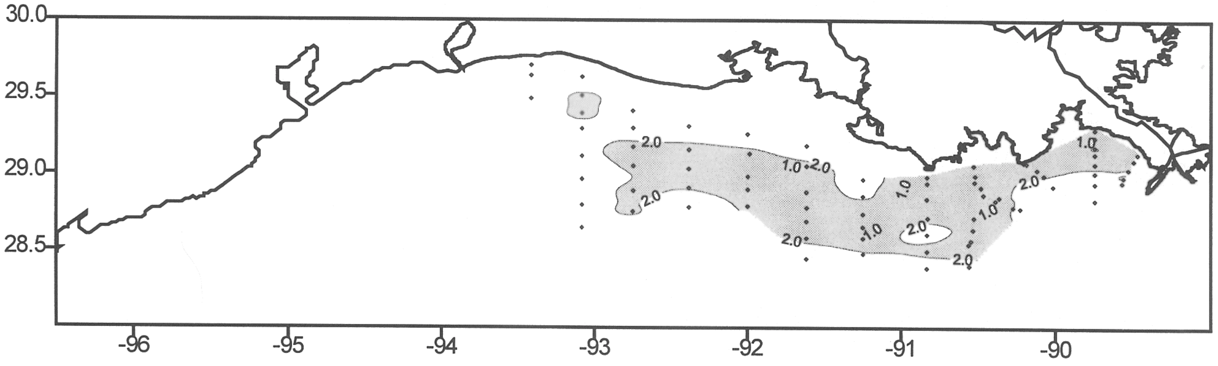
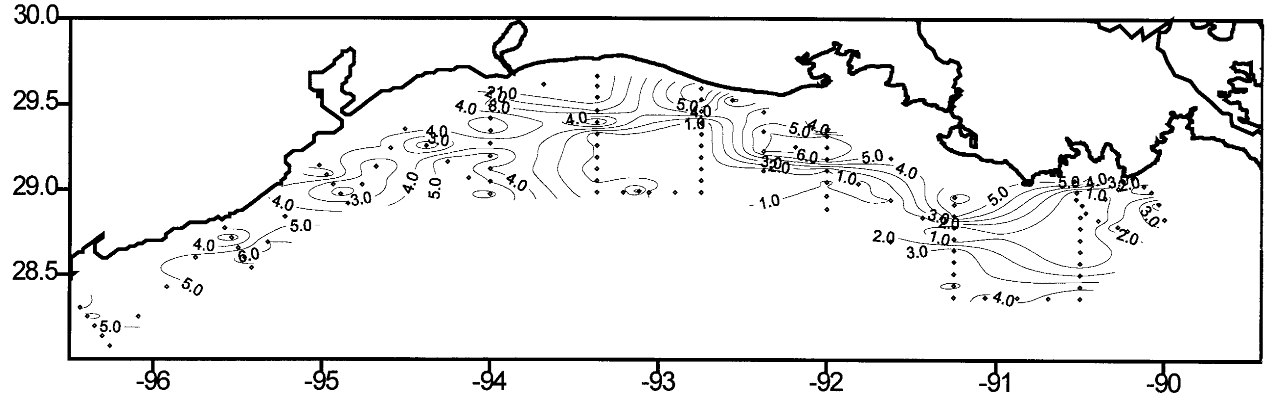


FIGURE 3.5. Areal extent of hypoxia ($\leq$ 2 mg/l, stippled) for July 1994 LATEX and NECOP cruises. NOTE: Upper panel outlines the two study areas. (*Modified from Rabalais 1998b.)*

LATEX July 12-18, 1994
Bottom Dissolved Oxygen (mg/L)



NECOP July 24-29, 1994
Bottom Dissolved Oxygen (mg/L)

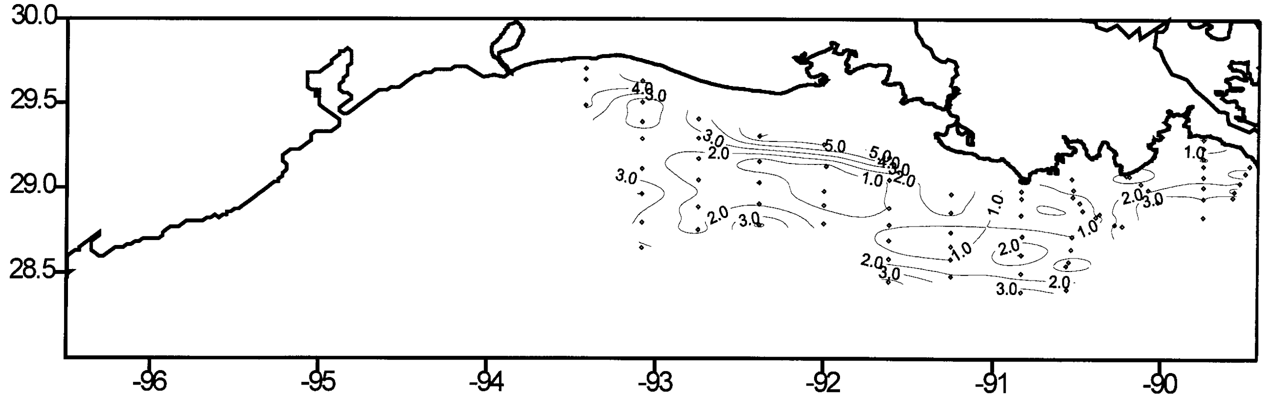
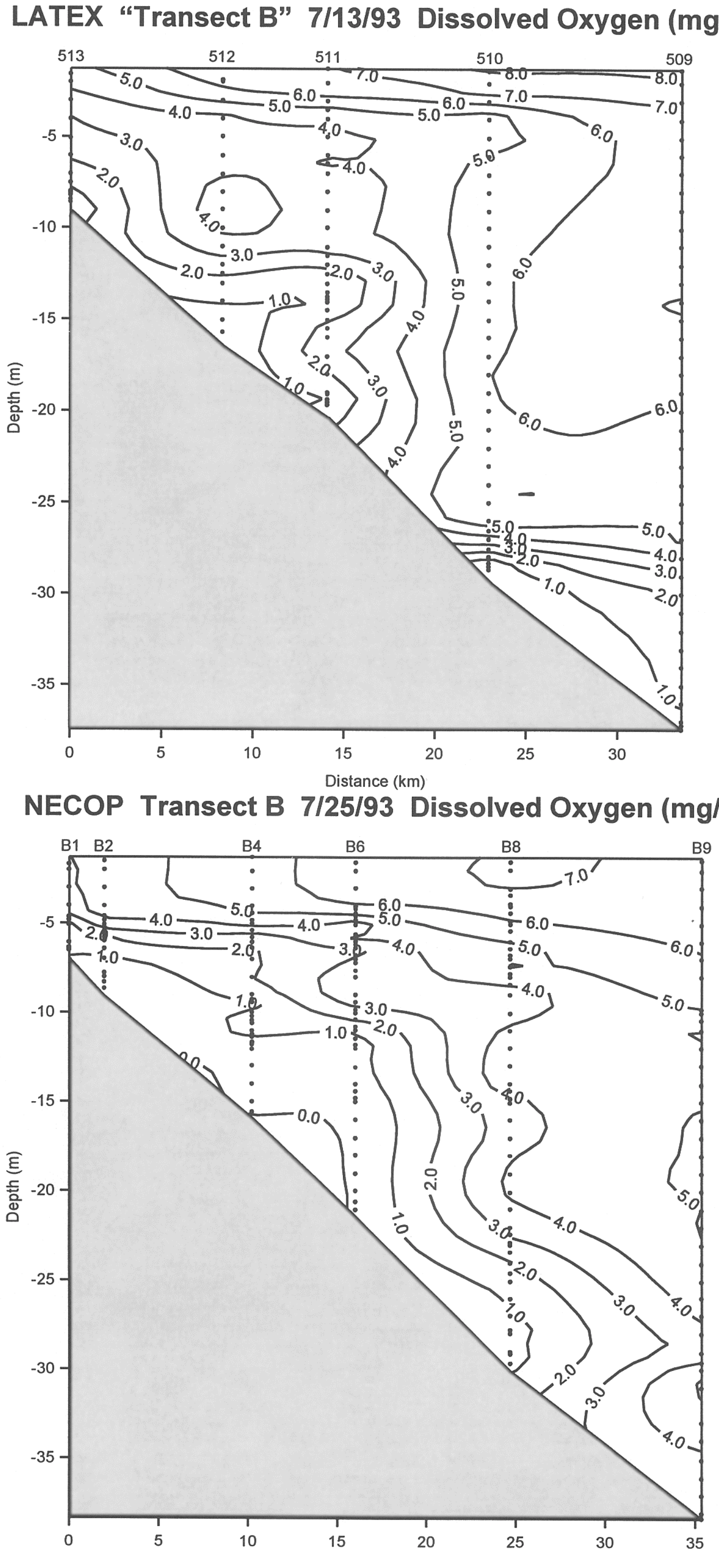


FIGURE 3.6. Bottom-water dissolved oxygen for July 1994 LATEX and NECOP cruises. (*Modified from Rabalais 1998b.)*


LATEX “Transect B” 7/13/93 Dissolved Oxygen (mg/L)

NECOP Transect B 7/25/93 Dissolved Oxygen (mg/L

FIGURE 3.7. Cross-shelf contours (transect B) for dissolved oxygen for July 1993 LATEX and NECOP cruises as indicated. (*From Rabalais 1998b.)*

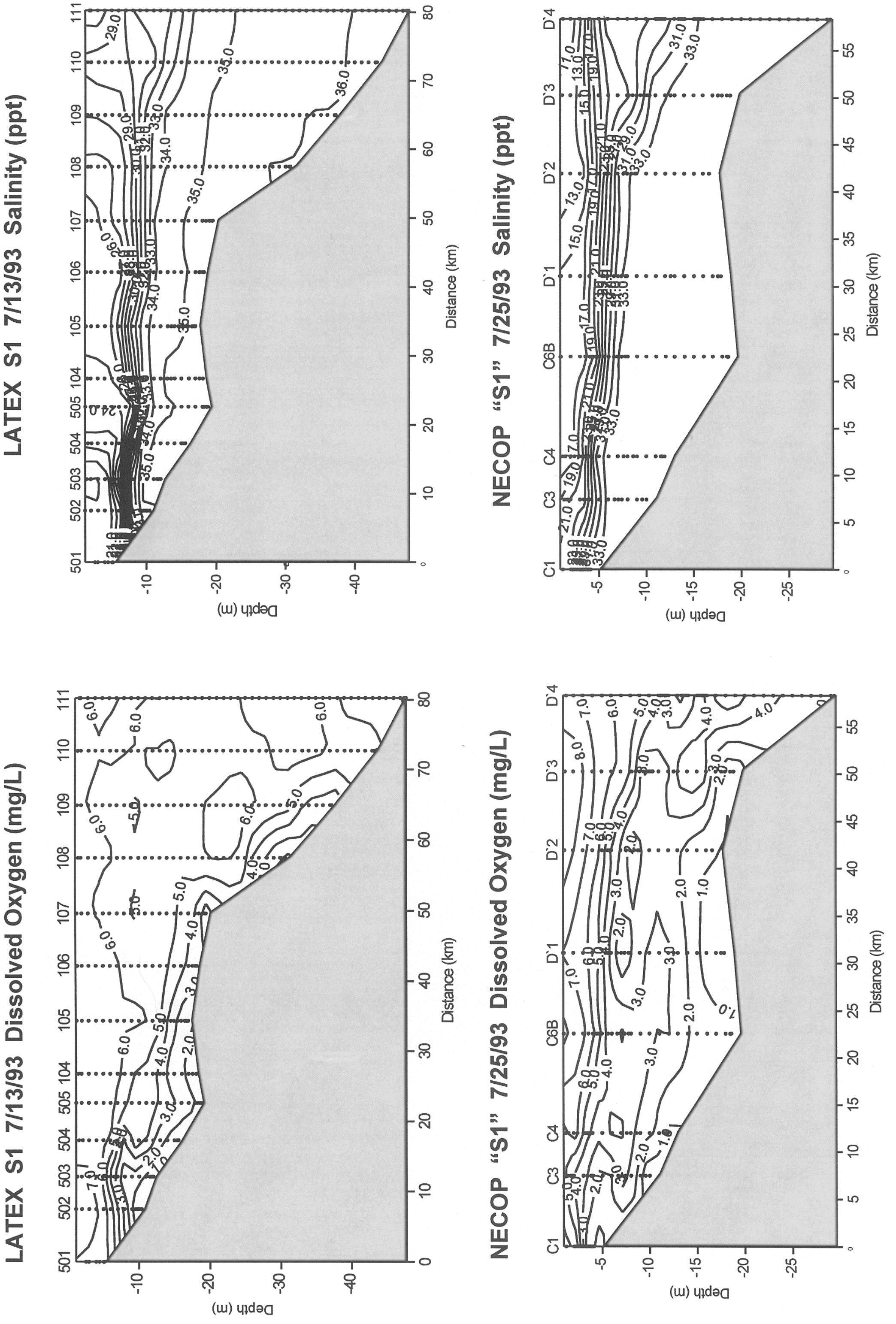
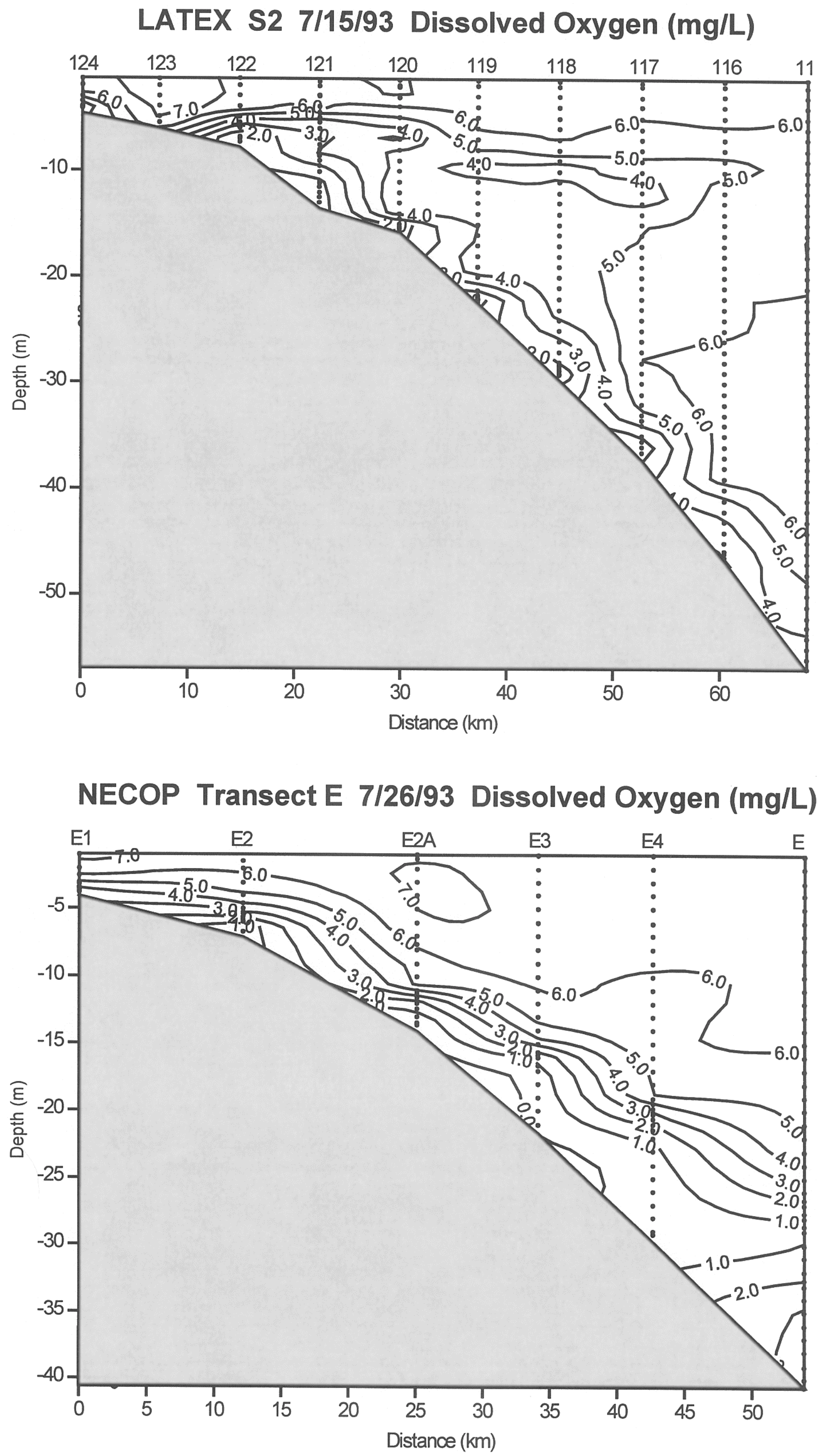


FIGURE 3.8. Cross-shelf contours (S1 line) for dissolved oxygen and salinity for July 1993 LATEX and NECOP cruises as indicated. (*From Rabalais 1998b.)*



LATEX S2 7/15/93 Dissolved Oxygen (mg/L)

NECOP Transect E 7/26/93 Dissolved Oxygen (mg/L)

FIGURE 3.9. Cross-shelf contours (S2 line and transect E) for dissolved oxygen for July 1993 LATEX and NECOP cruises as indicated. (*From Rabalais 1998b.)*

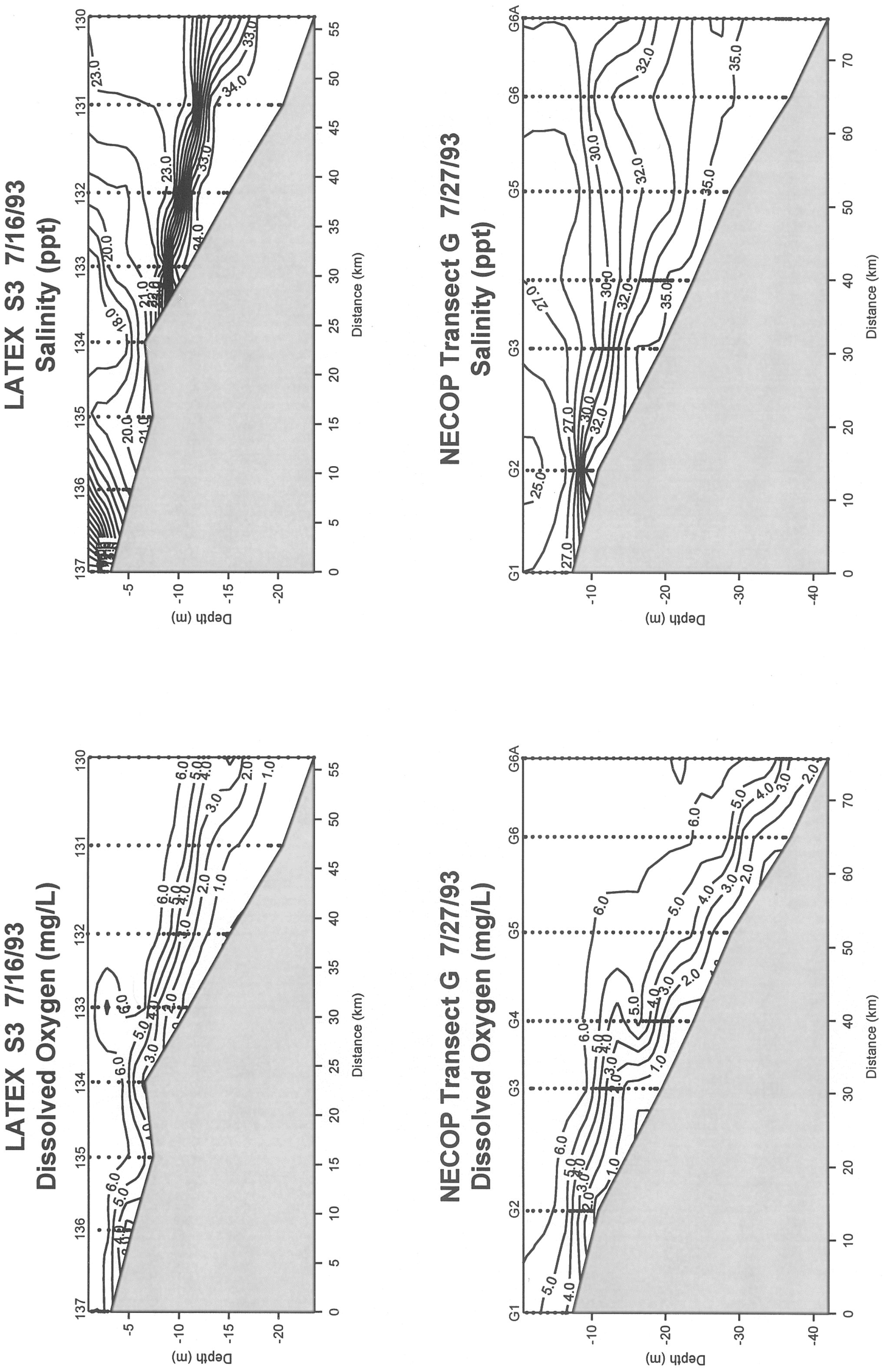


FIGURE 3.10. Cross-shelf contours (S3 line and transect G) for dissolved oxygen and salinity for July 1993 LATEX and NECOP cruises as indicated. (*From Rabalais 1998b.)*

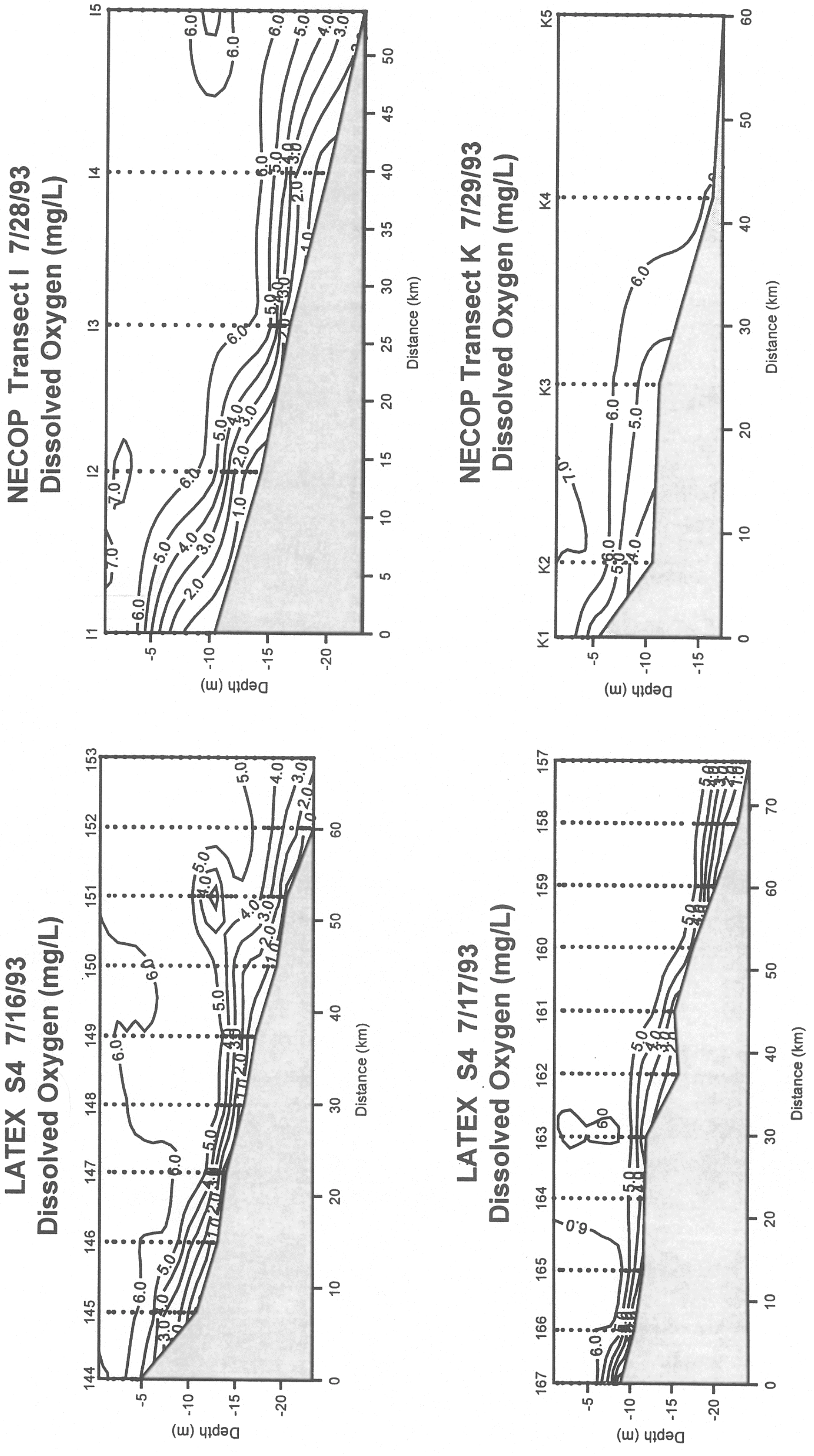


FIGURE 3.11. Cross-shelf contours (S4 line and transect I; S5 line and transect K) for dissolved oxygen for July 1993 LATEX and NECOP cruises as indicated. (*From Rabalais 1998b.)*

The oxygen concentration was slightly below 2 mg/l in the near-bottom waters at only one station off Terrebonne Bay during the LATEX April 1992 cruise (Figure 3.12). However, there was an area of oxygen concentration approaching hypoxia in a band along the mid- to outer stations of the S1 and S2 lines from Terrebonne Bay to Point au Fer Island, respectively. Isolated areas of near-bottom waters low in oxygen but not hypoxic (2 < x < 4 mg/l) were located off the Sabine and Galveston estuaries. The low-oxygen conditions were confined to the lower 2 m of the water column, compared to a much thicker layer of oxygen deficiency in mid-summer. Comparable shelf-wide NECOP data were not available for April 1992, but a cross-shelf comparison between the LATEX S1 line (4/22/92) and NECOP transect C (4/10–11/92) indicated the persistence of a mid-transect oxygen-depletion feature on both dates and a similar density structure.

Hypoxic near-bottom waters were not documented in the LATEX 13–22 April 1993 cruise (Figure 3.12). Oxygen-deficient, but not hypoxic, waters (2 < x < 4 mg/l) were located along the S1, S2, and S3 lines from offshore Terrebonne Bay to offshore Atchafalaya Bay (Figures 3.12 and 3.13). Lower oxygen concentrations were present along most of the length of the S1 line in the lowest 2 to 3 m of the water column. The remainder of the LATEX study area was generally well aerated.

In contrast, several distinct patches of hypoxic bottom waters were observed during the NECOP 26–30 April 1993 cruise the following week that were not evident during the prior LATEX cruise (Figure 3.14). The distribution of the low-oxygen area off Terrebonne Bay (NECOP, 27 April) was consistent with the lower concentrations of dissolved oxygen along the S1 line in mid-April (LATEX, 13 April) (Figure 3.13). Reduced winds and calm seas were conducive to the further development of hypoxia off Terrebonne Bay during the interim.

The minimal distribution of oxygen-depleted bottom waters during April 1992 and 1993 is consistent with the long-term database for transect C off Terrebonne Bay (see Sections 3.4 and 3.5). Mississippi River discharge was unusually low in April 1992 (approximately 15,000 m^3/s), compared to normal high spring discharge in April 1993 (30,000 m^3/s). The surface salinity field was fresher in the region of the S1 and S2 lines in April 1993 than in April 1992, and the density stratification was stronger. Less oxygen deficiency in April 1992 compared to 1993 may be related to reduced flow, less stratification, nutrient flux, and surface net production, but may also result from the passage of two cold fronts during that cruise, compared to moderate southeasterly winds off Louisiana during April 1993 (i.e., less likely to disrupt the density structure). Continuous oxygen measurements (Appendix C) from 1992 vs. 1993 indicated that development of hypoxia in 1992 was frequently disrupted by mixing events every 7–10 days. In 1993 a gradual decline in oxygen throughout the whole month of April indicated the lack of mixing events.

There are no shelf-wide data for the months of May, June, or mid- to late August.

A cruise on 10–13 September 1985 followed a series of hurricanes on August 14 and 29. Stratification had redeveloped in the Mississippi River Bight, and a single station was hypoxic (Figure 3.15). The remainder of the stations were well mixed and well aerated.

A single shelf-wide LATEX cruise in October 1992 documented a well-mixed water column and well-oxygenated waters throughout the column (Figure 3.12). The period prior to and during the cruise was characterized by strong winds and rough seas. These results were consistent with the long-term database for transect C off Terrebonne Bay (see Sections 3.4 and 3.5). Hypoxia has not been recorded later than the first week of October and is not likely to occur after strong cold fronts move across the area and/or thermal cooling destabilizes the density structure.

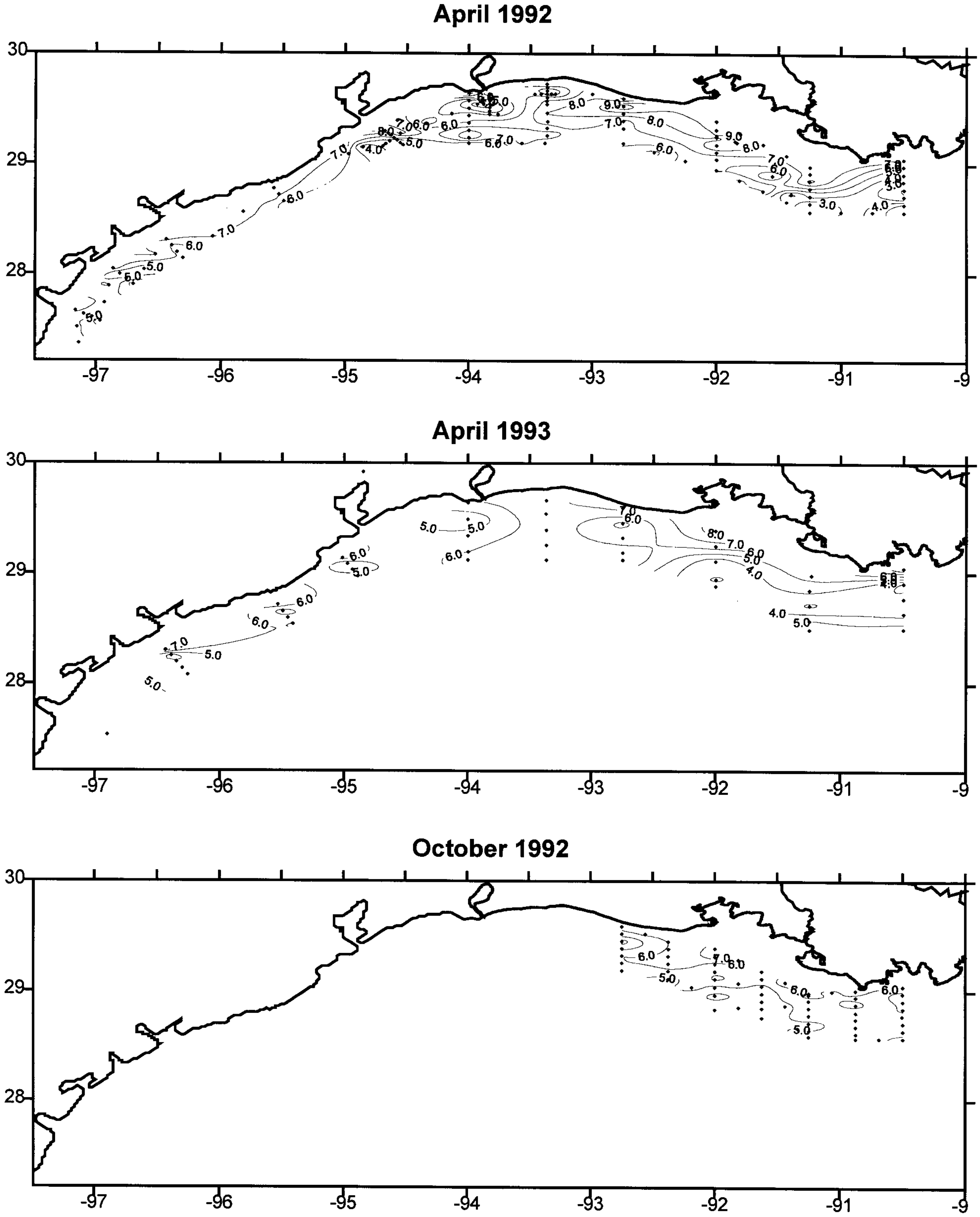


FIGURE 3.12. Bottom-water dissolved oxygen for April and October LATEX cruises as indicated. (*From Rabalais 1998b.)*

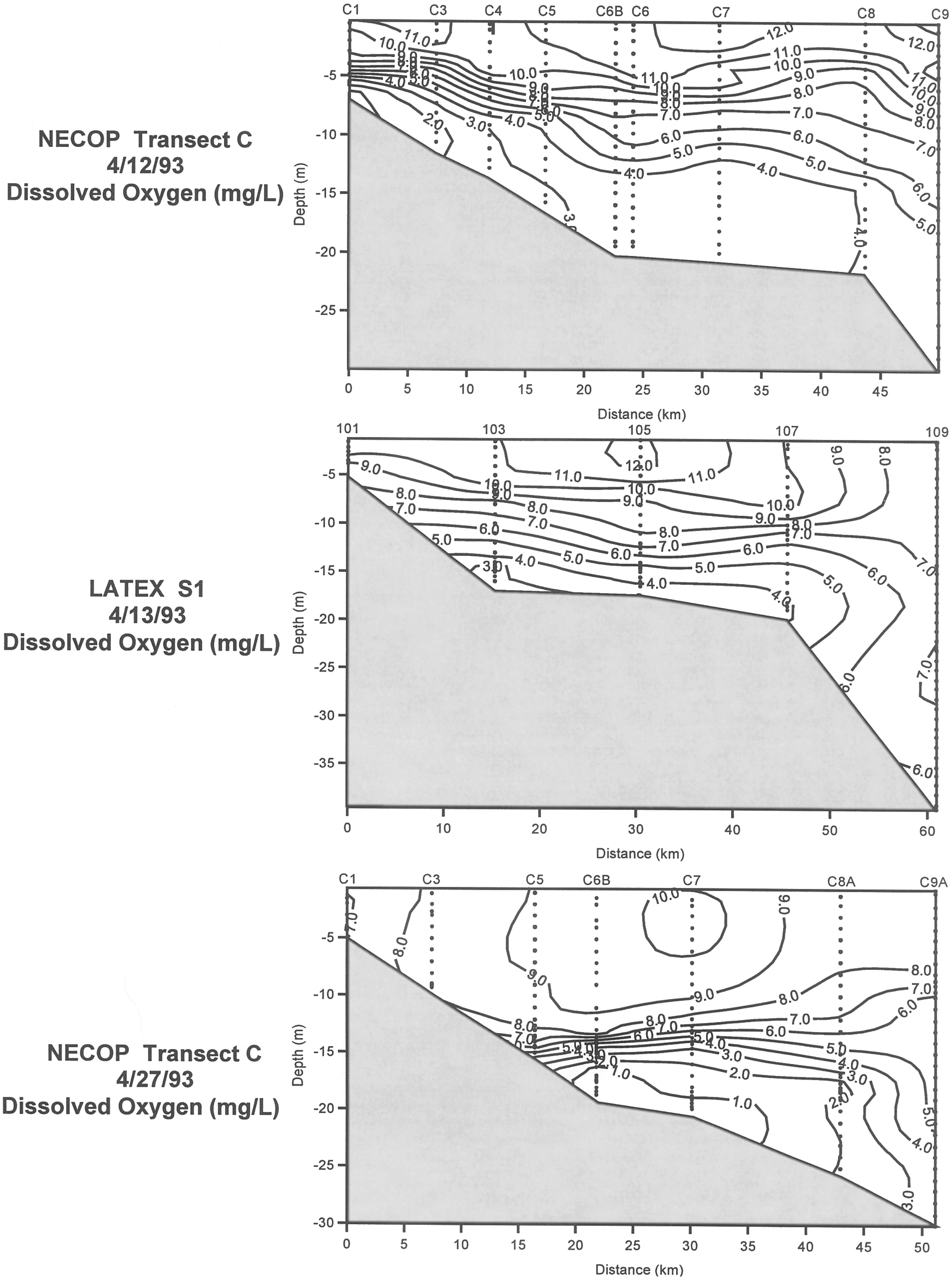


FIGURE 3.13. Cross-shelf contours for dissolved oxygen for April 1993 LATEX and NECOP cruises as indicated. (*From Rabalais 1998b.)*

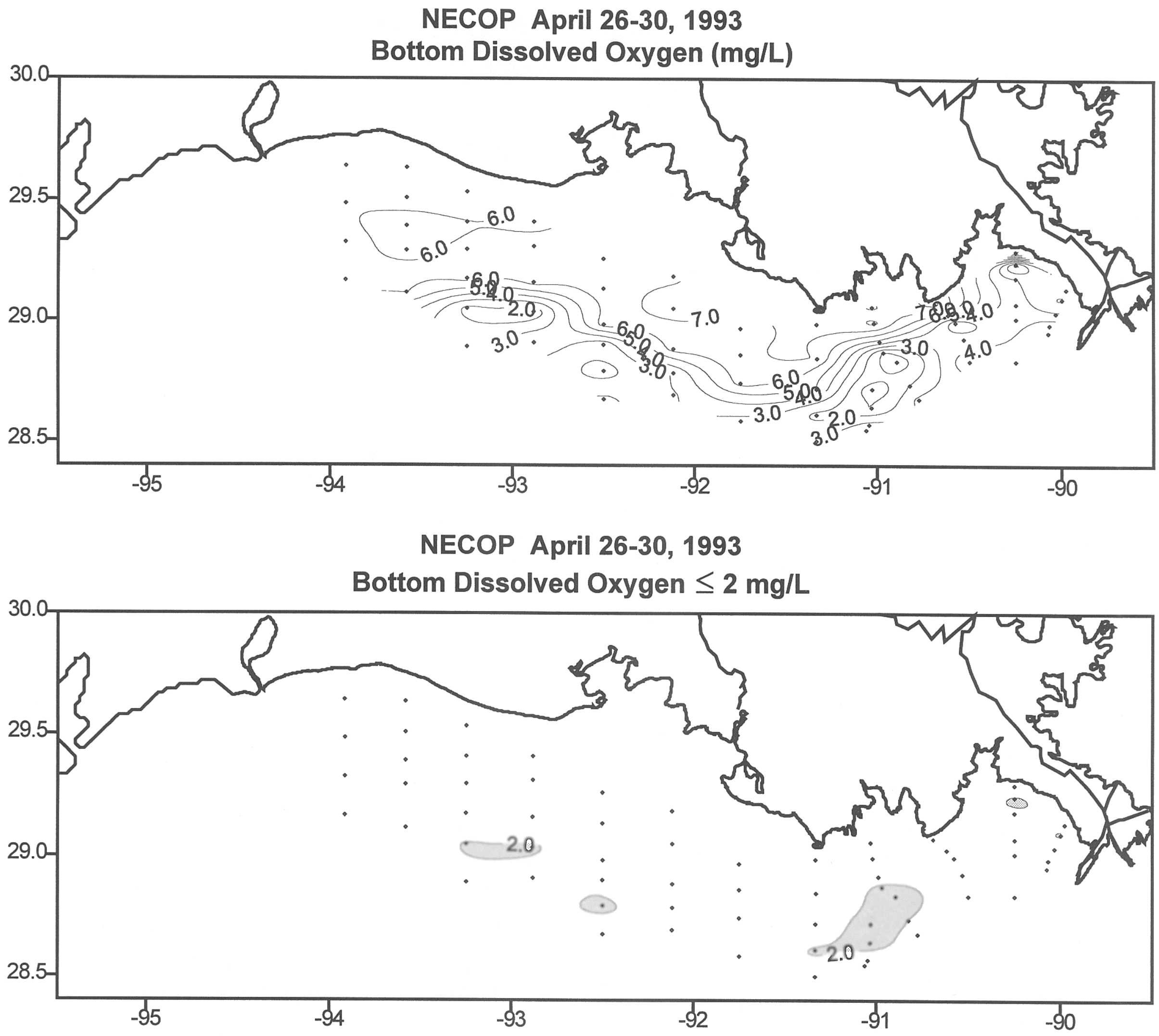
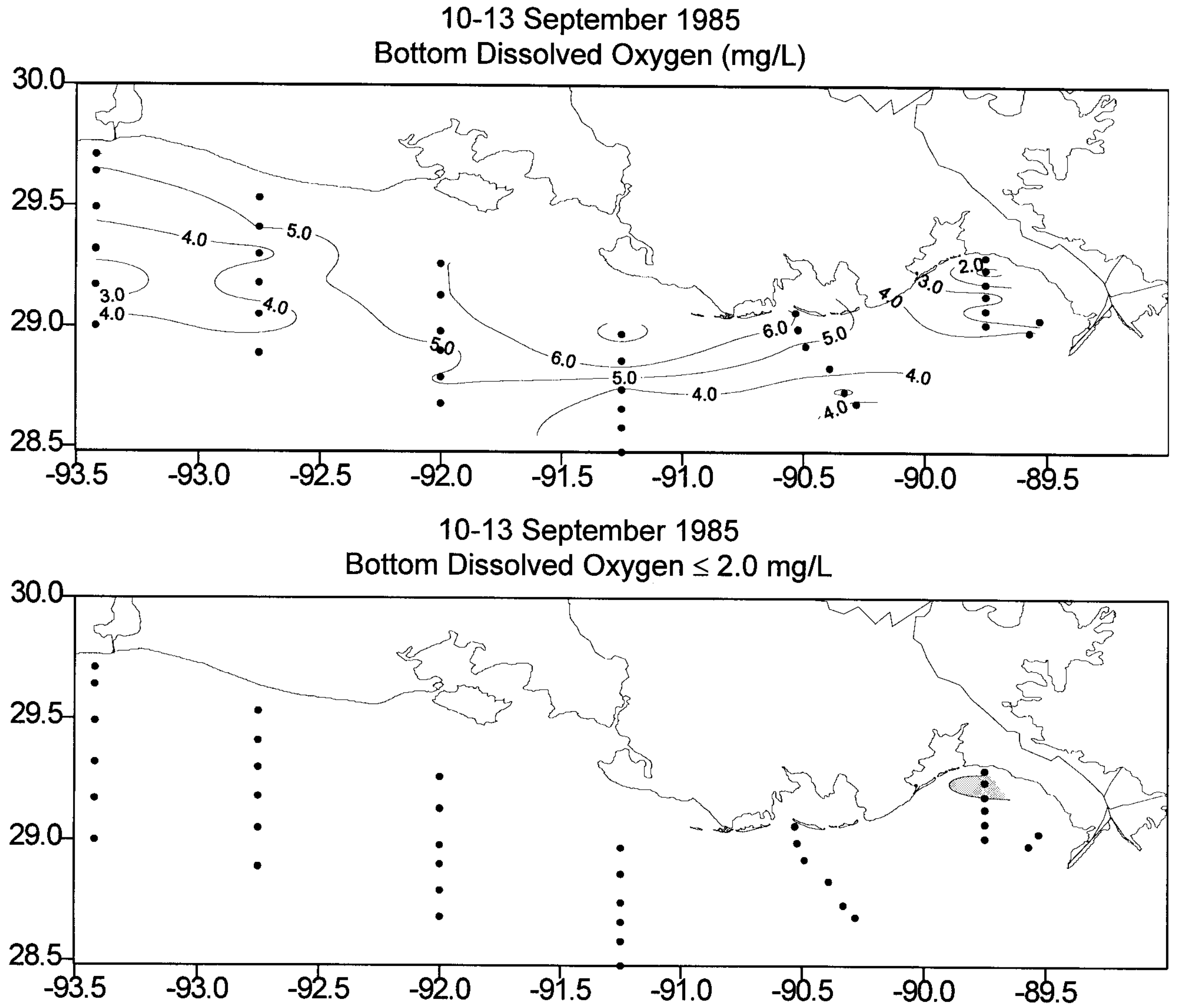


FIGURE 3.14. Bottom-water dissolved oxygen for April 1993 NECOP shelf-wide cruise (upper panel) and areal extent of hypoxia (≤ 2 mg/l, stippled) (lower panel). (*From Rabalais et al. unpublished data.)*


10-13 September 1985
Bottom Dissolved Oxygen (mg/L)

10-13 September 1985
Bottom Dissolved Oxygen ≤ 2.0 mg/L

FIGURE 3.15. Bottom-water dissolved oxygen for September 1985 and areal extent of hypoxia (≤ 2 mg/l). (*Drawn from data in Rabalais et al. 1986.)*

3.4 CROSS-SHELF SPATIAL AND TEMPORAL VARIABILITY, SOUTHEASTERN LOUISIANA

The maps of the mid-summer extent of hypoxia provide a benchmark for yearly comparisons, even though they are minimal estimates of the total seabed area subjected to hypoxia and not necessarily representative of conditions throughout the summer. Also, the persistence of these broad areas is unknown except for limited data in 1993–94 (Section 3.2), because sequential mapping cruises have not been conducted throughout the year. Better temporal data that define the extent, persistence, and important biological and physical processes are available from the southeastern Louisiana shelf, where monthly data have been collected since 1985 and continuously recording oxygen meters were deployed near the bottom in water 20 m deep (Section 3.5).

Critically depressed dissolved oxygen concentrations occur below the pycnocline from as early as late February through early October and nearly continuously from mid-May through mid-September. In March, April, and May hypoxia occurs across less of the seabed along transect C and is not continuous. Hypoxia is most widespread, persistent, and severe in June, July, and August (Rabalais et al. 1991). The persistence of extensive and severe hypoxia into September and October depends primarily on the breakdown of the stratification by winds from either tropical storm activity or passage of cold fronts. Hypoxia is rare in the late fall and winter.

Once hypoxia occurs, much of the onshore–offshore variability, especially in mid-summer, can be attributed to wind-induced cross-shelf advection. Cross-shelf oxygen isopleths in Figure 3.16 and Appendix B illustrate the changes in oxygen contours in response to different wind conditions. The low-oxygen water mass is displaced into deeper water under downwelling favorable conditions. Upwelling favorable conditions push the hypoxic mass closer to the barrier shore.

Comparisons of the cross-shelf transects demonstrate that hypoxia is more widespread and severe in some years than in others (for transect C on the southeastern shelf). Hypoxia developed as early as April in 1986 and persisted through mid-October, and was extensive across the bottom and well up into the water column for most of the spring and summer (Figure 3.16). Similarly extensive and severe low oxygen was observed in 1990, 1992, 1993, 1995, and 1996 but not in 1991 and 1994 (Appendix B); data for 1997 and 1998 are not yet compiled.

3.5 TIME SERIES

Continuously recording (15-min intervals) oxygen meters were deployed near the bottom at Station C6A or C6B during the spring and fall of 1990–98. The mooring was also instrumented with near-surface and near-bottom current meters. (Time-series bottom-oxygen concentration data for 1989–97 are in Appendix C.) There is variability within and between years, but the patterns generally depict (1) gradual decline of bottom-oxygen concentrations through the spring, with reoxygenation from wind-mixing events; (2) persistent hypoxia and often anoxia for extended periods of the record in May–September; (3) isolated wind-mixing events in mid-summer that reaerate the water column, followed by a decline in oxygen similar to that seen in the spring; (4) isolated upwelling of higher oxygen-content waters from deeper water during upwelling favorable wind conditions, then a relaxation of the winds and a movement of the low-oxygen water mass back across the bottom at the site of the oxygen meter; and (5) wind-mixing events, either tropical storms/hurricanes or cold fronts in the late summer and fall that mix the water column sufficiently to prevent prolonged instances of dissolved oxygen concentrations less than 2 mg/l.

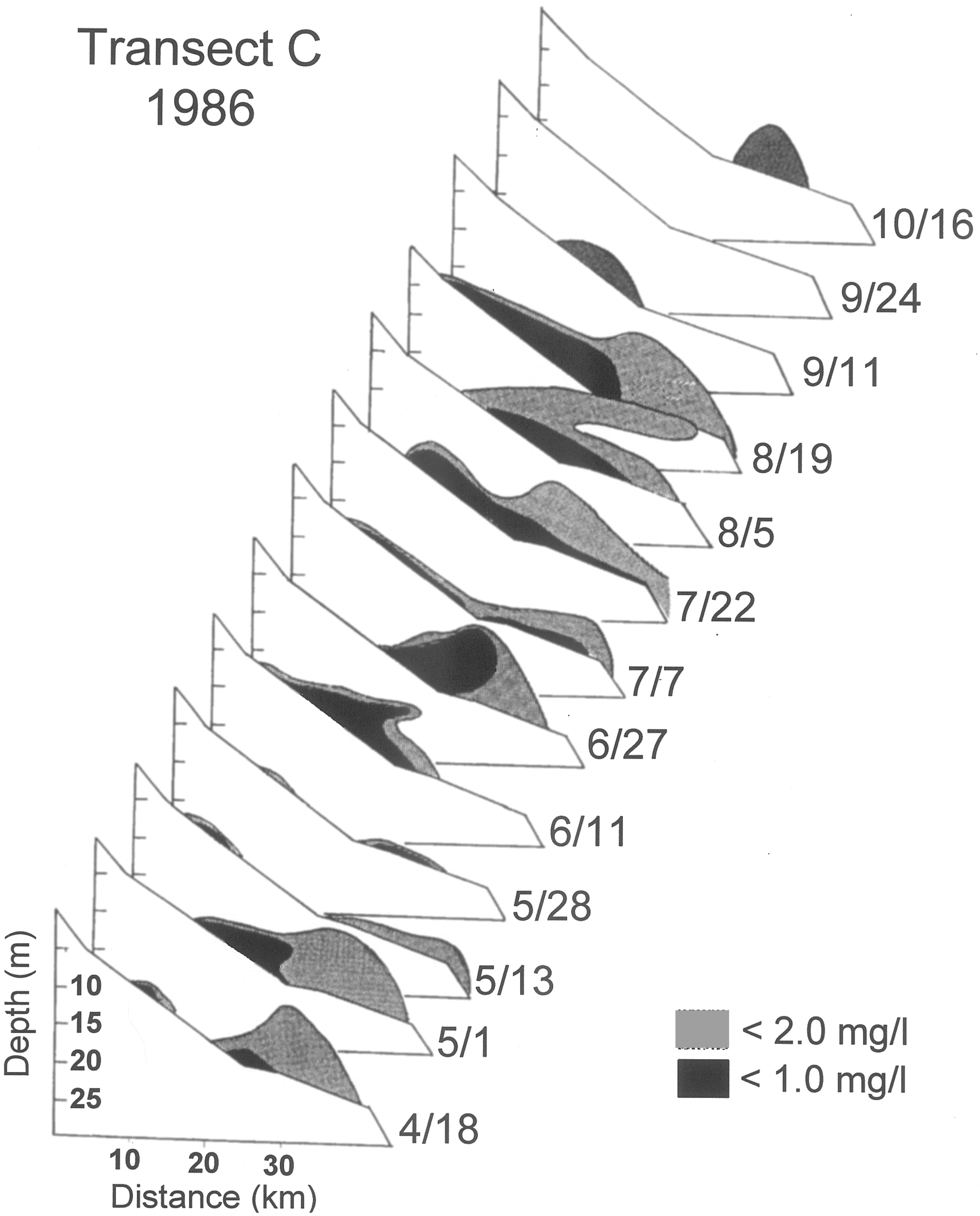


FIGURE 3.16. Cross-shelf (transect C) distribution of dissolved oxygen less than 2 mg/l (stippled) and less than 1 mg/l (black) for 1986. (*Drawn from data in Rabalais et al. 1986.)*

Comparative bottom-oxygen records were obtained from two locations 77 km apart in water 20 m deep (Rabalais et al. 1994), one of which was at the seasonally persistent hypoxic station C6A and illustrative of bottom oxygen conditions in the core of the hypoxic zone (Figure 3.17). The second station was closer to the Mississippi River Delta in the vicinity of station A3 (Figure 2.1). The oxygen meters recorded considerably different oxygen conditions for a four-month deployment from mid-June through mid-October. At station C6A, bottom waters were severely depleted in dissolved oxygen and often anoxic for most of the record from mid-June through mid-August, and there were no strong diurnal or diel patterns. At the station 77 km to the east and closer to the Mississippi River Delta (WD32E), hypoxia occurred for only 50% of the record, and there was a strong diurnal pattern in the oxygen time-series data. There was no statistically significant coherence between the oxygen time series at the two stations. Coherence of both oxygen records with wind records was weak. The dominant coherence identified was between the diurnal peaks in the WD32E oxygen record and the bottom-pressure record from a gauge located at the mouth of Terrebonne Bay. This suggested that the dissolved oxygen signal at WD32E was due principally to advection of the interface between hypoxic and normoxic water by tidal currents.

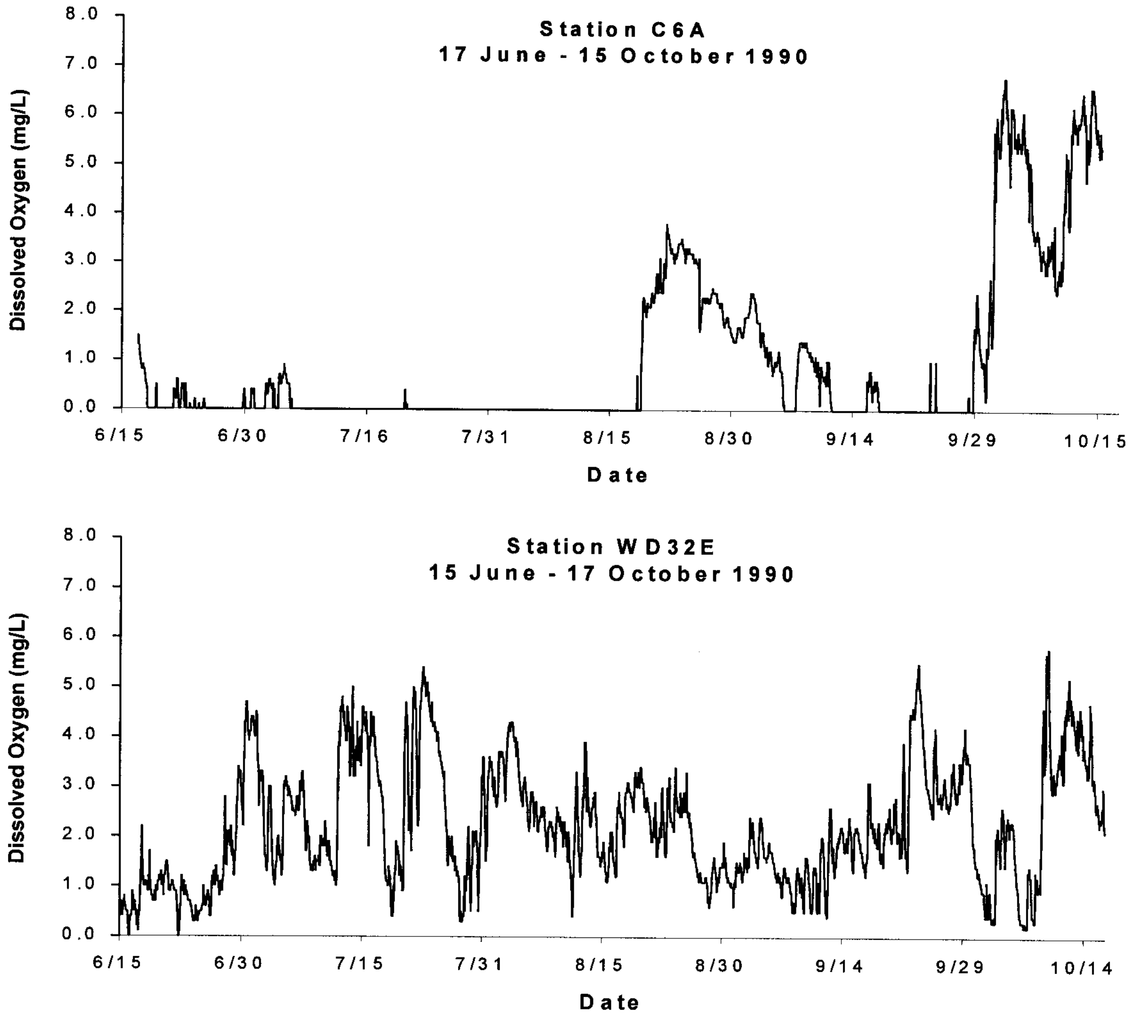


FIGURE 3.17. Time-series plots of near-bottom dissolved oxygen concentration (mg/l in 1-h intervals) at stations C6A and WD32E. (*From Rabalais et al. 1994.)*

3.6 AVERAGE CONDITIONS

The more frequent data collections at station C6* (combined data for C6A, C6, and C6B, see Figure 2.1) provide a description of seasonal changes in hypoxia and related physical and biological variables. The data from C6* have also allowed for the development of oxygen budgets and models and identification of changes over an annual cycle in relation to Mississippi River discharge and nutrient flux (see Sections 6.9 and 6.11). Data for surface and bottom waters for station C6* are illustrated in Figures 3.18 and 3.19, with the average conditions for 1985–92 compared to the deviations from the norm during the 1993 flood conditions. Data frequency is greater during March–October than during November–February.

Surface waters at station C6* reflect seasonal changes in the discharge of the Mississippi and Atchafalaya Rivers and seasonally variable currents that move the surface plume into the area. Surface salinity is normally 25–30 ppt and somewhat lower in late spring and summer. Bottom-water salinity averages 30–35 ppt, so that surface-to-bottom water differences and strength of the stratification are also greatest in late spring and summer. The concentration of dissolved inorganic nitrogen (NO_3^-, NO_2^-, and NH_4^+) is highest in the spring, and the makeup of the dissolved nitrogen pool is primarily NO_3^- in the spring, with a higher proportion of NH_4^+ in the summer. Dissolved orthophosphate PO_4^{3-} is generally higher in the spring, but quite variable. Dissolved silicate SiO_4^- is also quite variable through the year (indicative of riverine flux and uptake and regeneration by diatoms).

Surface chlorophyll *a* concentration as an indicator of phytoplankton biomass follows a seasonal trend similar to that of the dissolved inorganic nitrogen but lagged in time. Primary production is sustained through the summer and fall by regenerated nitrogen. Surface phaeopigment concentrations follow the same pattern as chlorophyll *a*, but at much lower values (approximately five times less). There is a seasonal progression of phytoplankton taxonomic groups, with diatoms being more abundant in the spring and picocyanobacteria peaking in the summer (see Sections 3.7 and 6.4). Bottom-water chlorophyll levels follow the spring peak in the surface waters, and bottom phaeopigment levels (an indicator of fluxed phytoplankton and fecal pellets) are highest in the summer. The seasonal cycle in bottom-water oxygen is well defined, with highest values in the fall and winter, a decline in the spring, and lowest values in the summer. The long-term monthly average bottom-water oxygen concentration falls below 2 mg/l in June–August.

Representative, average mid-summer conditions for selected stations in the core of the hypoxic zone were derived from the long-term averages at stations along the 20-m isobath (Figure 3.20). Mid-summer surface-water salinity is usually 25–30 ppt, with the values being somewhat lower on the southeastern shelf, especially adjacent to the Mississippi River Delta (transect A´). Coupled with long-term bottom salinity values of 35 ppt, the result is a strong salinity stratification across the shelf (at least within the 20-m isobath). Mid-summer surface-water nutrient values are usually low, with slightly higher values on transects near the Mississippi and Atchafalaya Deltas (an example of dissolved inorganic nitrogen appears in Figure 3.20). Similarly, surface chlorophyll *a* concentrations for the 20-m stations are generally less than 5 µg/l across the shelf, except adjacent to the Mississippi River Delta). Bottom-water dissolved oxygen levels at 20 m are lowest west of the Mississippi and Atchafalaya Rivers (see Figure 3.1).

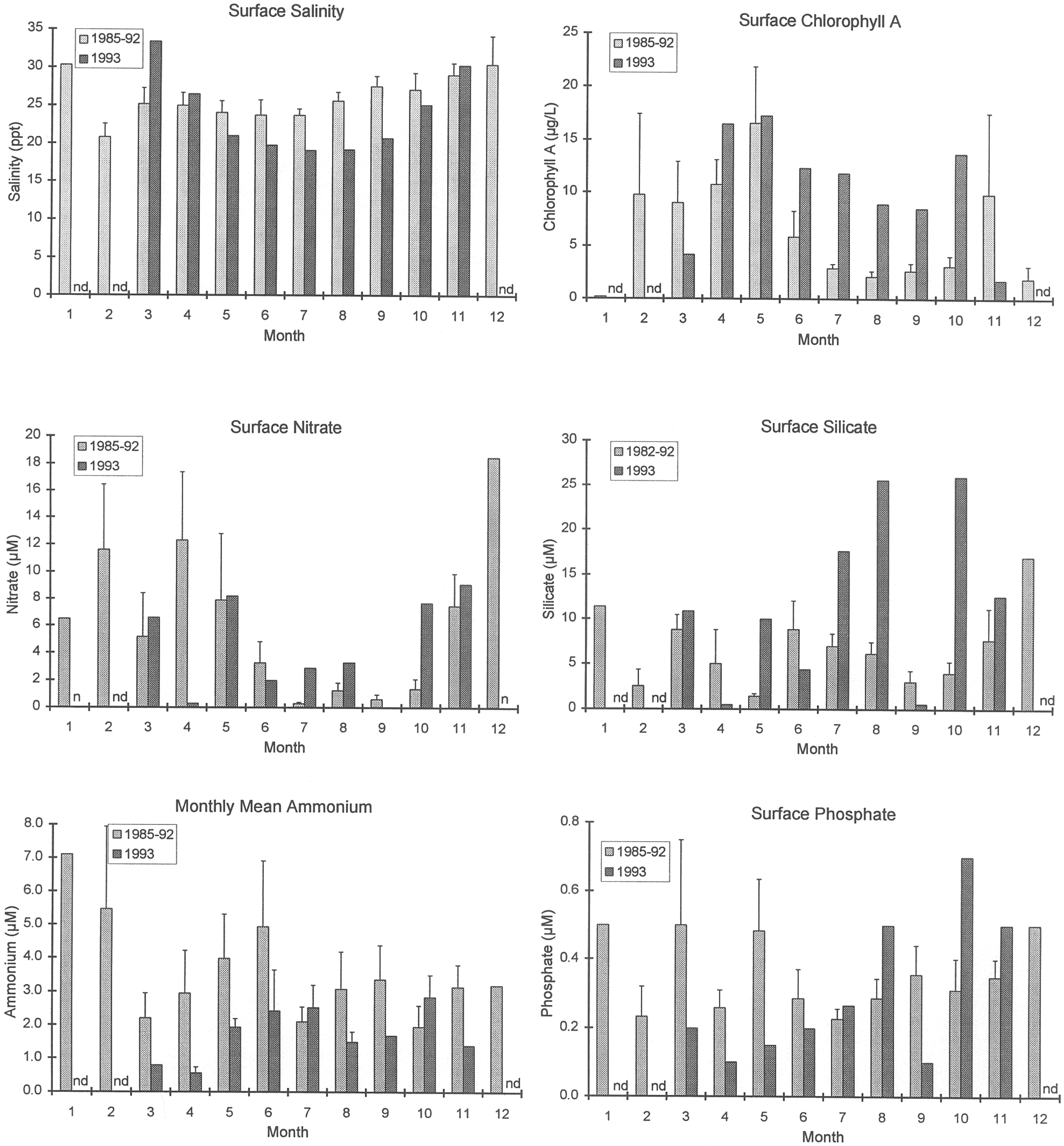


FIGURE 3.18. Surface-water quality for station C6* (composite data for Stations C6A, C6B, and C6) for 1985–92 average conditions (± s.e.) compared with 1993. NOTE: *n* for average condition ranges 10–30 in March–October and 2–6 in November–February. *n* for 1993 is 1 or 2. (*From Rabalais et al. 1998.)*

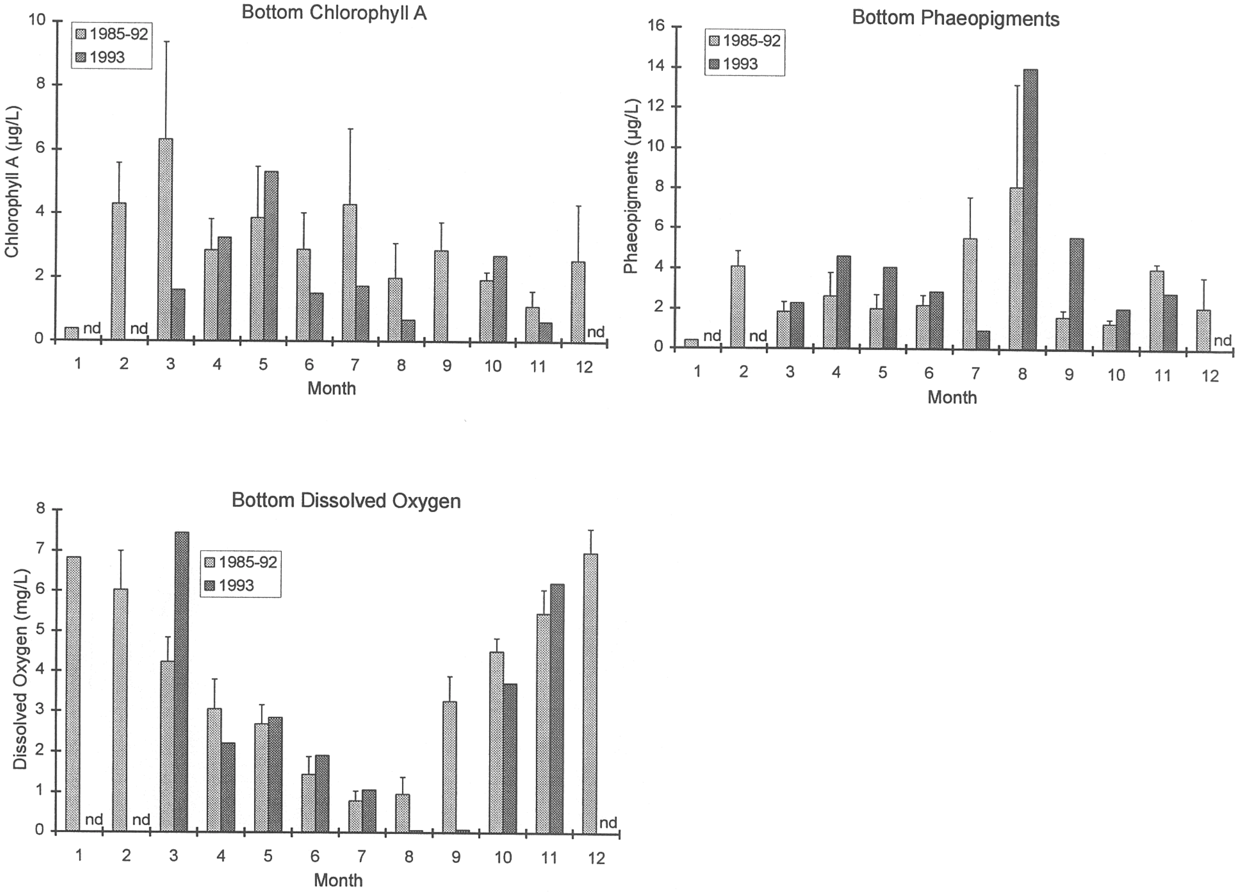


FIGURE 3.19. Bottom-water quality for station C6* (composite data for Stations C6A, C6B, and C6) for 1985–92 average conditions (± s.e.) compared with 1993. NOTE: *n* for average condition ranges 10–30 in March–October and 2–6 in November–February; *n* for 1993 is 1 or 2. (*From Rabalais et al. 1998.)*

3.7 EXTREMES IN RIVER DISCHARGE

Conditions during extreme events, such as the 1993 flood or the 1988 drought, can be compared to the long-term average conditions (see Section 3.6) or more "typical" flow years to identify factors that are important in influencing the distribution of hypoxia. These comparisons, however, cannot be isolated from the physical and biological conditions occurring weeks or months prior to those documented during a cruise, or from the physics of the system during the surveys.

The influence of Mississippi River system discharge and flux of nutrients was magnified during the 1993 flood (Dowgiallo 1994). Above-normal freshwater inflow and nutrient flux from the Mississippi and Atchafalaya Rivers from late spring well into mid-summer and early fall (Figure 3.21) were clearly related in time and space to the seasonal progression of hypoxic water formation and maintenance and its increased severity and areal extent on the Louisiana–Texas shelf in 1993 (Rabalais et al. 1998). Long-term average flow of the Mississippi River (as measured at Tarbert Landing, Mississippi, downstream of the Old River Control Structure where 30% of the flow is diverted into the Atchafalaya River) exceeded both the long-term averages for March–July and the long-term maximum daily record between August 5 and September 10 (Boyles and Humphries 1994). Flow from mid-September through December continued well above the long-term averages. The higher-than-average flows during late spring continued through the summer and early fall of 1993. Several researchers documented either higher concentrations of riverine nutrients or flux of nutrients to the Gulf of Mexico in 1993 (Goolsby et al. 1993; Goolsby 1994; Whitledge 1994; Justić et al. 1997; Rabalais et al. 1998). Nitrate flux during the spring of 1993 was two times higher than the previous eight-year average and three- and fourfold higher in July and August, respectively, than the long-term average (Justić et al. 1997).

Surface-water salinity at station C6B was much lower during May–October 1993 than the average value for the previous eight years (Figure 3.18). Bottom-water salinity was similar to the long-term average and resulted in greater surface-to-bottom salinity differences during May–October in 1993. Nitrate concentrations in surface waters that are normally elevated in spring were also elevated in the spring of 1993 and continued at higher-than-normal levels through October (exception in September). Total dissolved inorganic nitrogen in surface waters in 1993 paralleled normal elevated levels in spring and additionally high values in October 1993. Silicate concentrations were well above the long-term average in July, August, and October 1993. Phosphate levels in 1993 were generally lower than the long-term spring averages, but higher than the summer and fall averages, except in September.

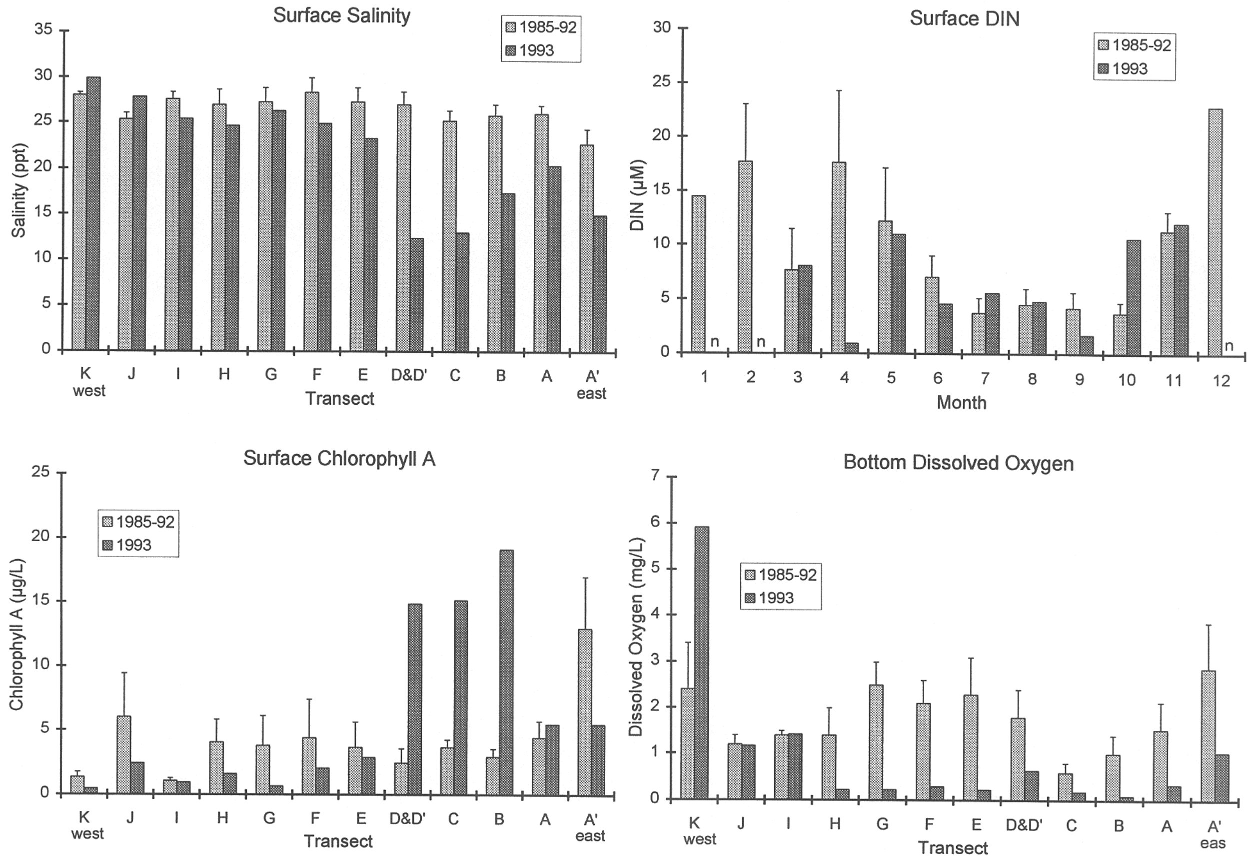


FIGURE 3.20. Surface- and bottom-water quality data for 20-m depth stations on each transect from the Mississippi River (Transect A') to the upper Texas coast (Transect K) for the long-term average conditions of 1985–92 (± s.e.) compared with 1993 (stations shown in Figure 2.1. NOTE: *n* for the average conditions ranges 6–9; *n* for 1993 is 1 or 2 (resampling of station C6B). *(From Rabalais et al. 1998.)*

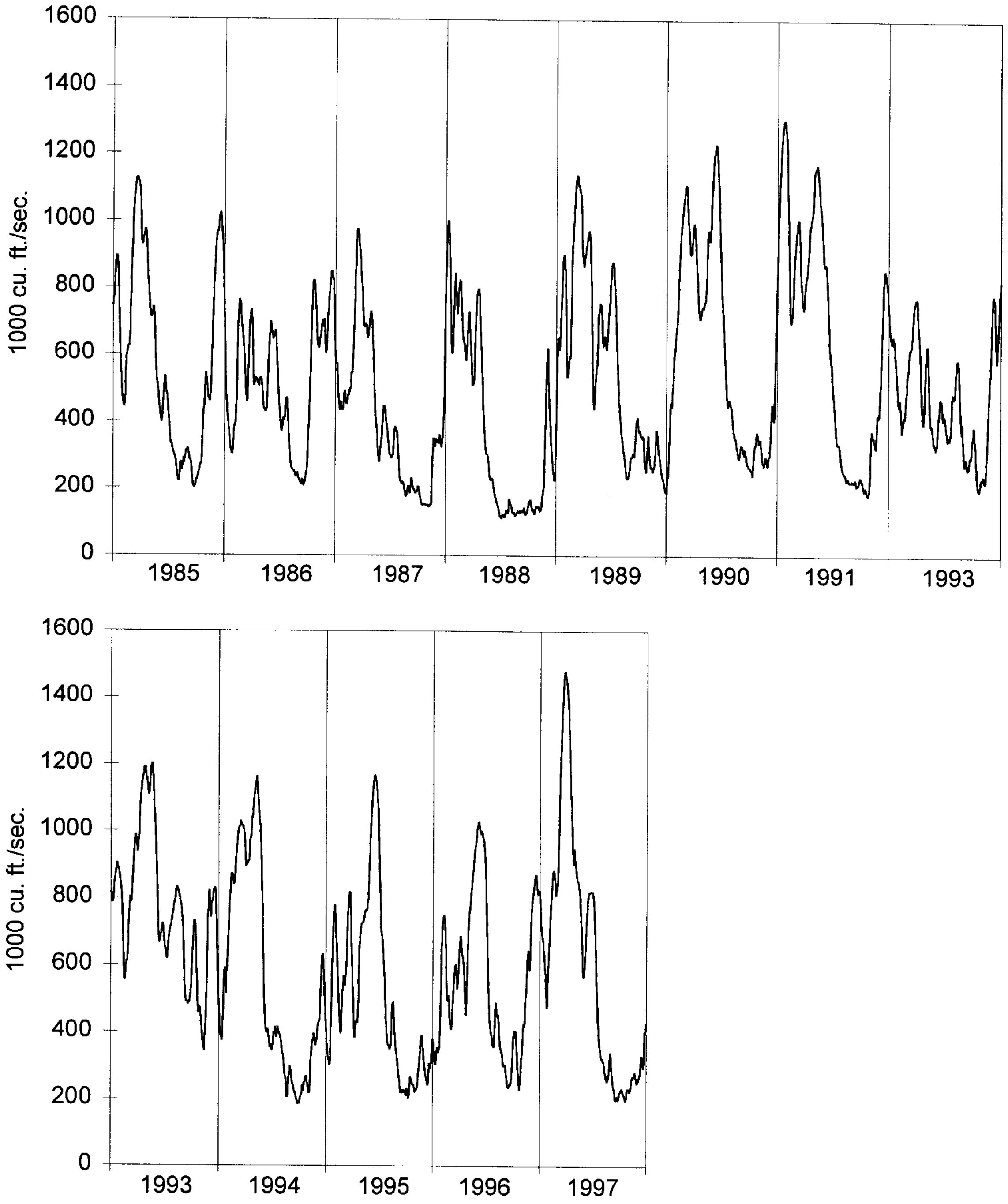


FIGURE 3.21. Daily discharge of the Mississippi River at Tarbert Landing. (*Data from U.S. Army Corps of Engineers.*)

Surface-water chlorophyll *a* concentrations peaked in April and May 1993, similar to the long-term average, but continued well above average during June–October (Figure 3.18). Phaeopigment concentrations in surface waters for 1993 were similar to the long-term average. Total phytoplankton numbers were approximately an order of magnitude greater in August–October 1993 than the 1990–92 average (Figure 3.22). Most of this increase in abundance was due to increases in small (1- to 2-μm), coccoid cyanobacteria. Summertime shelf phytoplankton abundance is typically dominated by cyanobacteria, but diatoms that are much larger (5–100 μm) usually dominate the biomass (see Section 6.4). During July–September 1993, however, estimated cyanobacterial biomass was two times greater than the estimated diatom biomass (Dortch 1994). Although cyanobacterial numbers and biomass increased the most, diatom numbers also increased threefold during July–September 1993 compared to the 1990–92 average. In contrast to the increases in cyanobacteria and diatoms, there was an order of magnitude decrease in dinoflagellate numbers. Chlorophyll *a* levels in bottom waters during June–September 1993 were lower than the long-term average. Bottom-water phaeopigment concentrations (as an indicator of fluxed degraded surface-water chlorophyll *a* biomass) were generally higher than the long-term averages for most spring, summer, and fall months in 1993. Bottom-water oxygen in 1993 followed the long-term seasonal decline through spring and summer until the peak in summer flooding in August and September, when near-anoxic conditions prevailed (Figure 3.19).

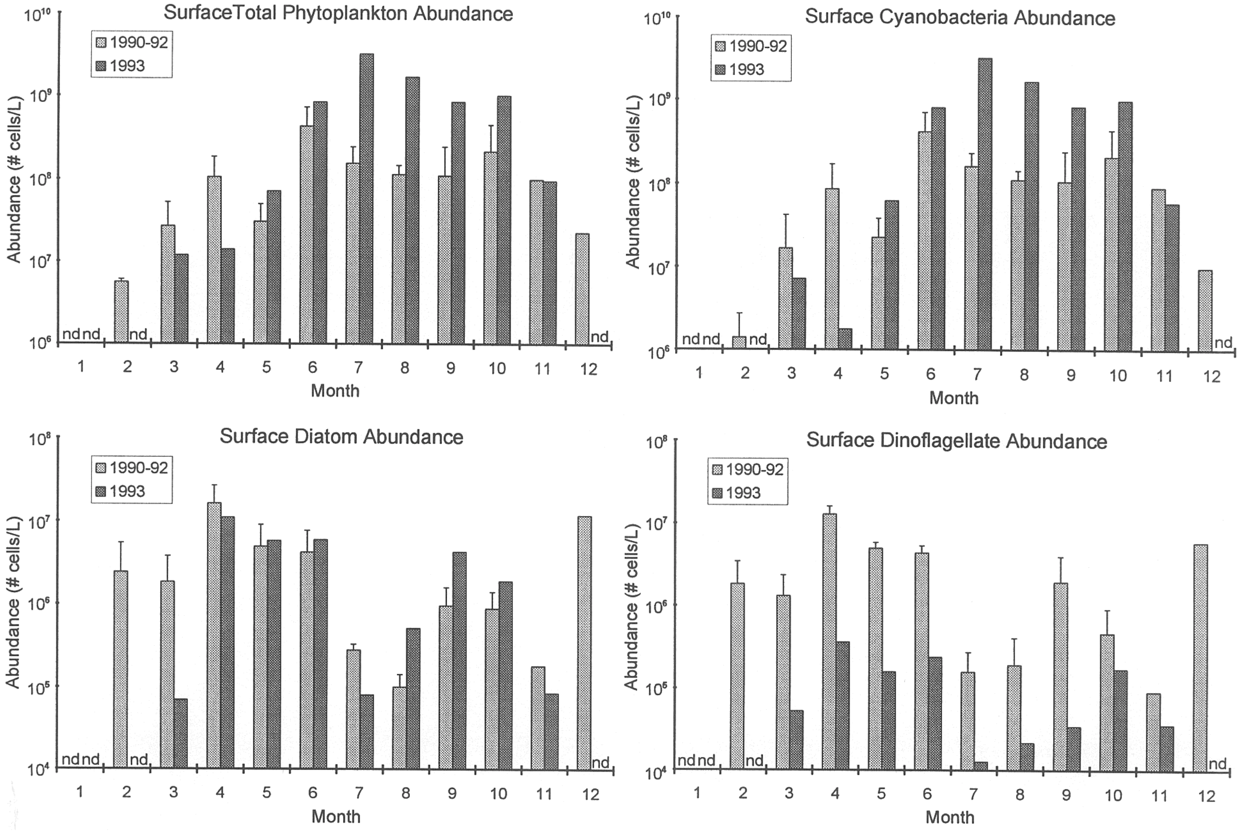


FIGURE 3.22. Comparisons of abundance of total phytoplankton, total cyanobacteria, total diatoms, and total dinoflagellates in 1993 at stations C6A or C6B compared to average data for 1991–92. NOTE: *n* for average conditions for most months is 3; *n* for 1993 is 1. *(From Rabalais et al. 1998.)*

Surface-water signatures in July 1993 of less saline, nutrient-rich, and high chlorophyll *a* biomass waters reflected the sustained and high freshwater outflow of the flooded Mississippi River system (Figure 3.20). Persistent southerly and southwesterly winds were typical for much of July through mid-August 1993 (Walker et al. 1994), causing the retention of large amounts of fresh water on the Louisiana shelf and upper Texas coast. Surface-water salinity along the 20-m isobath in 1993 was consistently and usually much lower than the long-term, mid-summer average, except on transects J and K to the west (Figure 3.20). Elevated concentrations compared to long-term averages were demonstrated for nitrate (transects A´-D), ammonium (transects A´-B), silicate (transects A´-D), and phosphate (transect A´). Substantially lower-than-average bottom-water dissolved oxygen values were found over most of the Louisiana inner shelf in 1993.

The size of the hypoxic zone in 1993 (estimated from NECOP data, 60- to 80-station grid) was two times greater in areal extent than the 1985–92 summer average (Figure 3.2). Results from three cruises in July 1993 that mapped bottom-water hypoxia (Figure 3.3) indicated that the hypoxic zone extended onto the upper Texas coast in early to mid-July, but was pushed back onto the Louisiana shelf in late July. These results are consistent with the winds and the acoustic Doppler current profiler data during mid- and late July (Murray 1998).

A 52-year low river flow of the Mississippi River occurred in 1988, and the response of the marine ecosystem was documented (Rabalais et al. 1991). Discharge began at normal levels in 1988 and quickly dropped to some of the lowest levels on record during the summer months (Figure 3.21). Normal flow in August is 8,000 m^3/s; but from June through August 1988, flow dropped to 4,000 m^3/s. In early June, hydrographic conditions on the southeastern Louisiana shelf were similar to those observed in previous years—i.e., a stratified water column and some areas of oxygen-deficient bottom waters. By mid-July, few areas of lower surface salinity were apparent, there was little density stratification, and low-oxygen conditions were virtually absent. Reduced summer flows in 1988 also resulted in reduced suspended sediment loads and increased water clarity across the continental shelf. The critical depth for photosynthesis was well below the depth of the seabed (in depths of 15–30 m), and photosynthetic production of oxygen in bottom waters was likely. A typical seasonal sequence of nutrient-enhanced primary production and flux of organic matter progressed in the spring and led to the formation of hypoxia. However, with the lack of stratification and the addition of oxygen in bottom waters via photosynthesis, hypoxia was not maintained.

The difference in hypoxia in a drought versus a flood year suggests a relationship between river discharge and the extent of hypoxia. A simple linear regression of mid-summer area and mean Mississippi River discharge for the preceding year produced an R^2 of 0.934 for nine years of data from 1985 to 1993 (Wiseman et al. 1997). This relationship, however, fails to hold for the additional years of 1994–98. Thus, a comparison of mid-summer area (a minimal and rough estimate) versus discharge is not entirely satisfactory. There is evidence, for example, that carbon burial in 1993 was sufficient to support the extensive 1994 hypoxic zone despite "normal" flow conditions (Justić et al. 1997). There is variability in the mid-summer extent of hypoxia due to physical conditions at the time of sampling—for example, the reduced mid-summer size in 1997 and 1998 due to the passage of a hurricane on the southeastern portion of the study area and the current regime, respectively. Stronger relationships are evident for hypoxia versus river discharge and nutrient flux with time and spatial lags (see Sections 6.2 and 6.11). A needed analysis is the regression (cross-correlation with time lags) of more precise volume estimates of shelf-wide hypoxia in mid-summer and for transect C through the year with river discharge and nutrient flux.

CHAPTER 4

Physical Oceanography

Two principal factors lead to the development and maintenance of hypoxia: a physically stratified water column and decomposition of fluxed organic matter. The physics of the system define where hypoxia can occur, and the biological processes of carbon production, flux, and respiration lead to oxygen depletion. The relative influence of these factors varies among environments with hypoxia, and within an environment over an annual cycle. In the northern Gulf of Mexico the two are complexly interrelated and directly linked with the dynamics of the Mississippi and Atchafalaya River discharge. The physical features of the system (described below) and the biological processes (described in Chapter 6) cannot be separated from each other and must be considered within the overall context of a large river interacting with a coastal sea.

Hypoxia in the northern Gulf of Mexico occurs on the inner to mid-continental shelf, rather than in a confined bight, enclosed sea, or estuary. The physical features that influence it are, therefore, of a shelf-wide scale. Also, the river flow that influences nutrient levels and freshwater inputs is large. The Mississippi River and its distributaries contribute 580 km^3/yr of fresh water to the Gulf of Mexico, along with sediment yields of 210 x 10^6 t/yr, 1.6 x 10^6 t/yr nitrate, 0.1 x 10^6 t/yr phosphorus, and 2.1 x 10^6 t/yr silica (Milliman and Meade 1983; Goolsby et al. 1999). The Mississippi River system discharge is a dominant factor in the physical oceanography and ecology of the northern Gulf.

4.1 PLUMES AND COASTAL CIRCULATION

The Mississippi and Atchafalaya Rivers are the primary riverine sources of fresh water to the Louisiana continental shelf (Dinnel and Wiseman 1986) and to the Gulf of Mexico. Combined, they account for 80% of freshwater inflow from U.S. rivers to the Gulf (Dunn 1996). The discharge of the Mississippi River system is controlled such that 30% flows seaward through the Atchafalaya River Delta, and 70% through the Mississippi River Delta. The former enters through two outlets into Atchafalaya Bay, a broad shallow embayment; the latter enters the Gulf through multiple outlets, some in deep water and some in shallow water. Approximately 53% of the Mississippi River Delta discharge flows westward onto the Louisiana shelf (U.S. Army Corps of Engineers 1974; Dinnel and Wiseman 1986), and the general flow of the Atchafalaya River effluent is to the west. The amount of water flowing eastward onto the Mississippi–Alabama shelf is unclear. Mississippi River influence, however, may be seen as far east as the Atlantic seaboard, under combinations of major floods and appropriate oceanographic conditions (Atkinson and Wallace 1975; Tester and Atkinson 1994; Walker et al. 1994).

The less dense, fresh-river discharge floats atop and mixes with the ambient coastal sea water. Initially, water enters the shelf as a buoyant plume near the river mouth. The plumes from Atchafalaya Bay and Southwest Pass of the Mississippi River Delta (and possibly other outlets of the Mississippi Delta) turn anticyclonically until they encounter the Louisiana coast (Wiseman et al. 1975). At this point they merge into the highly stratified Louisiana Coastal Current, often referred to as the "extended plume." The current flows westward along the Louisiana coast much of the year. The buoyant, low-salinity waters are separated from the waters of the mid-shelf region by a strong surface-to-bottom frontal zone that typically intersects the bottom near the 10–15 m isobath. This depth is somewhat deeper than expected from theory, because the winds are generally downwelling-favorable in the region and tend to force the lighter water against the coast. The significant depths of water (30–80 m) over which the Mississippi River plume flows for most of its journey before attaching to the coast tend to preclude the development of hypoxia in all but the worst conditions. Once merging into the Louisiana Coastal Current, the nutrient-laden, buoyant waters respond to a different set of dynamics.

Cochrane and Kelly (1986) described the seasonal mean circulation and surface salinity patterns within the Louisiana Coastal Current. Down-coast flow occurs throughout most of the year in response to both the buoyancy forcing from river discharge and the winds that are generally from the east in spring, fall, and winter over the Louisiana–Texas shelf (Gutierrez de Valasco and Winant 1996). A narrow band of low-salinity surface water extends from the Mississippi River Delta to, at least, the Texas–Mexican border (Smith 1980). Winds along the south Texas coast are up-coast in summer and push water northward and eastward toward the Louisiana shelf. A nearshore convergence zone occurs and low-salinity waters flow offshore as a jet (Murray et al. 1998). This convergence zone is believed to migrate up the coast until mid-summer. The flow reversal can extend as far east as Terrebonne Bay (Kimsey and Temple 1963, 1964; Figure 4.1).

By late summer, the winds relax and return to a more southeasterly and easterly flow over the Texas shelf, and the low-salinity surface band near the coast again extends into Texas waters. Hurricanes and tropical storms affect the hypoxic region during the summer and fall. The strong stirring and increased currents associated with these storms often vertically homogenize the water column, but stratification quickly returns following the storm. During the winter, the dominant synoptic weather pattern is cold-air outbreaks that affect the region every three to ten days (DiMego et al. 1976), blowing the waters offshore and up the coast. Cold-front activity is minimal during the summer.

The salinity signals associated with pulses of river discharge appear to travel westward along the coast at significant speeds (Wiseman and Kelly 1994). If the highly energetic, wind-driven signal is filtered from consideration, the cross-shelf dynamical balance at periods longer than ten days appears to be in geostrophic balance (Wiseman et al. 1997). The swiftest currents observed, however, are associated with storm winds. Except when under the direct influence of a hurricane, these winds are the result of cold-air outbreaks. Current speeds can exceed 50 cm/s in the surface layers. During the winter, similar speeds are observed in the near-bottom waters (Crout et al. 1984). In the stratified season, however, long-term-averaged, near-bottom speeds at station C6B in 20 m suggest a mean speed of 1 cm/s to the west, although instantaneous velocities can be significantly higher (Figure 4.2).

Besides mass and buoyancy, the effluent plumes from the Mississippi River carry suspended sediments and nutrients that are important to the productivity of the shelf. The nutrients fuel phytoplankton growth, while the suspended sediments alter the available light field.

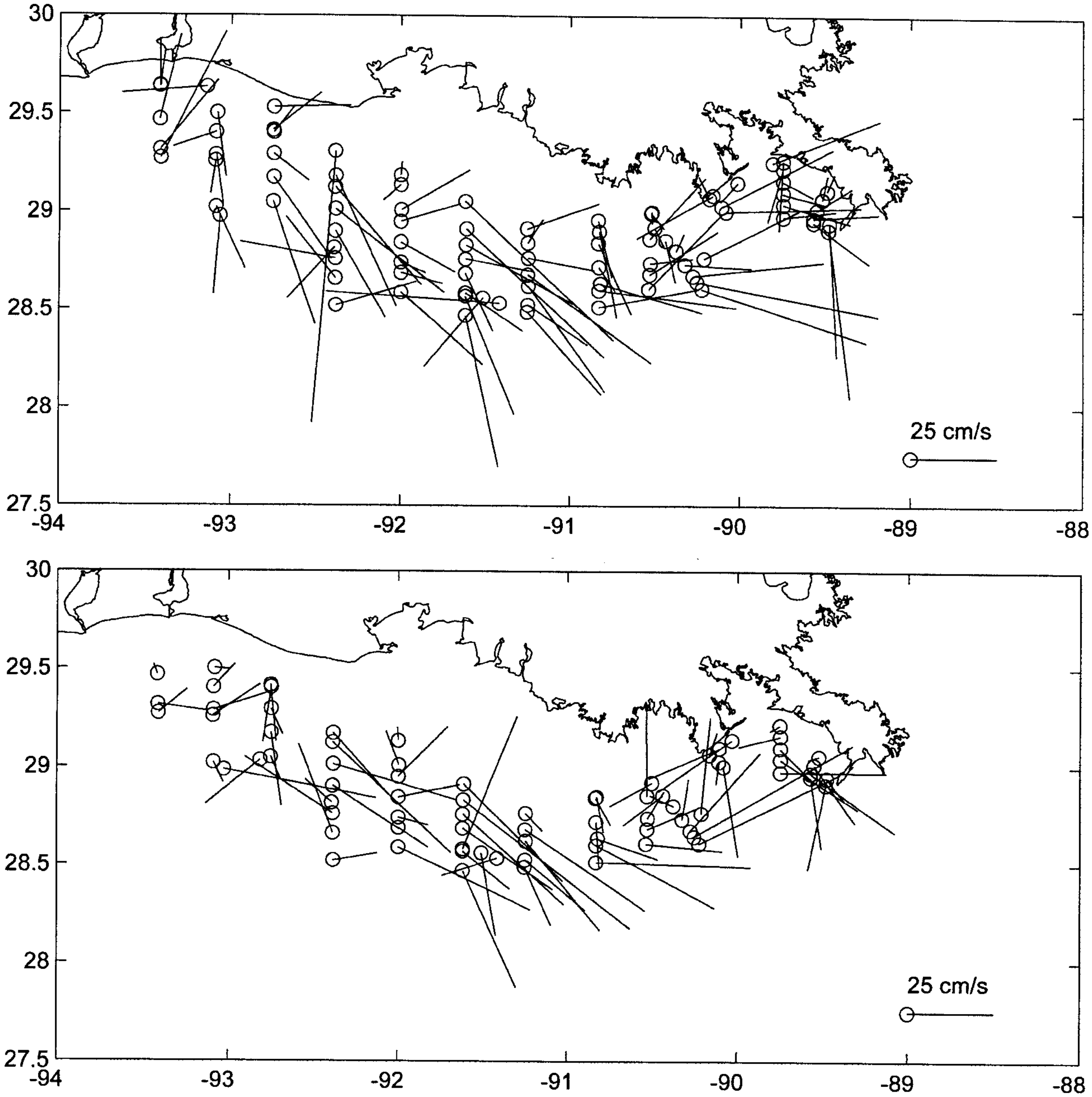


FIGURE 4.1. Acoustic Doppler current profiler data from mid-July 1993 at 4.7-m (upper panel) and 11.7-m (lower panel) depths in the water column. NOTE: The current direction is away from the open circle. (*From Wiseman et al. unpublished data*).

At the river mouth, suspended sediment concentrations in the plume are variable, but generally range from a few to a few hundred mg/l (Wright 1970; Walker 1996). The heavier grains settle rapidly from suspension, and only the fine silts and clays are carried greater distances. The associated suspended sediment can be observed in satellite images, and its variability has been used to map the spatial characteristics of the plume from Southwest Pass under differing conditions of discharge and wind (Walker 1996). The highest suspended sediment concentrations are observed relatively close to the river mouth, within a few tens of kilometers of the mouth, at most.

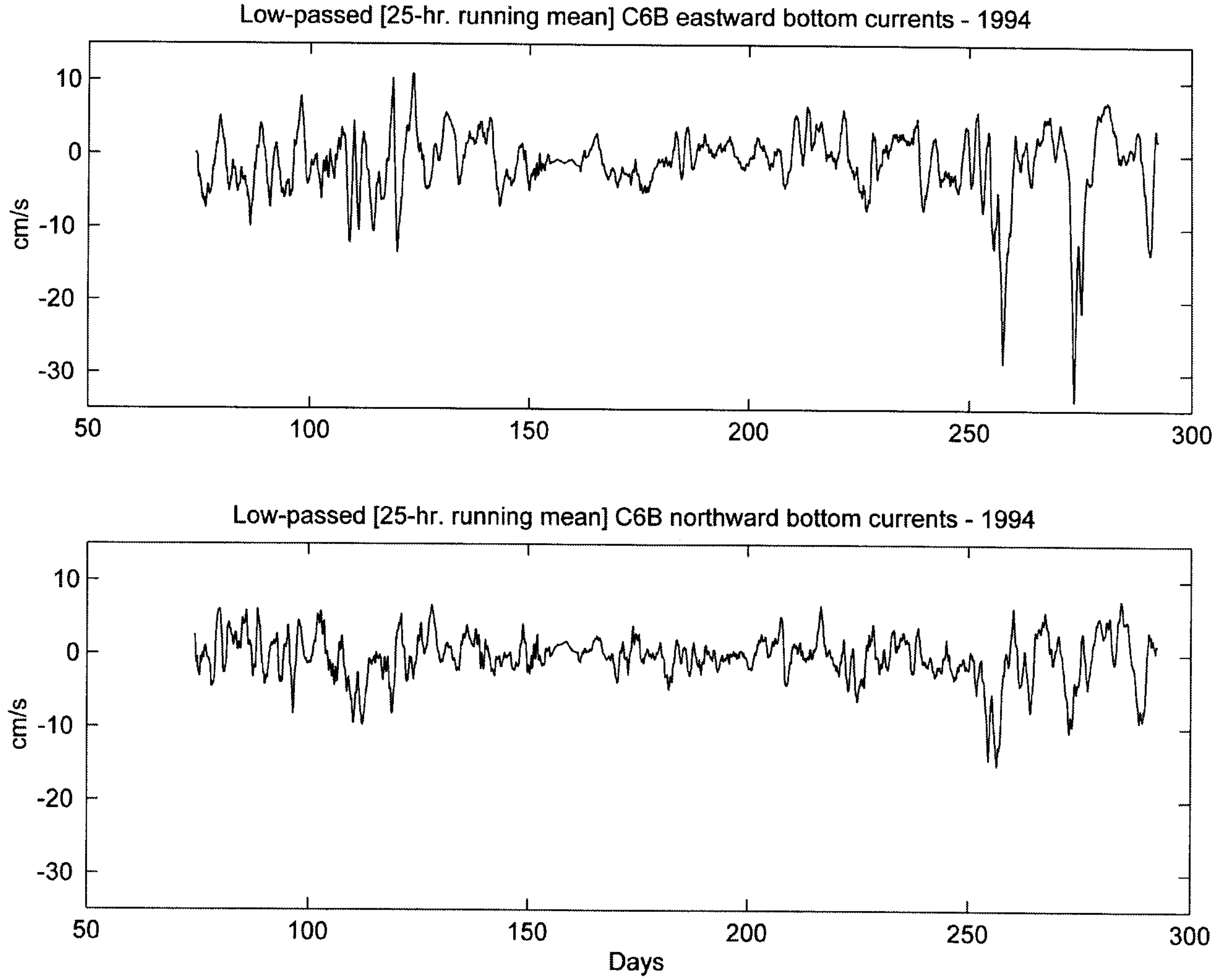


FIGURE 4.2. 1994 near-bottom current meter data from station C6B in 20-m water depth. NOTE: A 25-hour running mean filter has been applied to the data. The upper plot is eastward (along-shore) speed, and the lower plot is northward (cross-shore speed). (*From Wiseman et al. unpublished data*).

The general westward flow of the river effluent has an obvious effect on water quality west of the Mississippi River Delta. Data from numerous hydrographic cruises in all months between 85°W (just east of Cape San Blas, Florida) to 95°W (just west of Galveston Bay, Texas) in water 10–100 m deep were compiled by Rabalais et al. (1996) (Figure 4.3). Lower surface salinity values and shallower Secchi disk depths are apparent to the west of the Mississippi River Delta, with additional influences apparent near the Atchafalaya River Delta (92°W). A more pronounced decrease in concentration of silicate over an equal distance from the deltas when compared to the salinity plot indicates a nonconservative mixing due to biological uptake. Similar plots of nitrate and phosphate (not illustrated) indicated the same biological uptake processes.

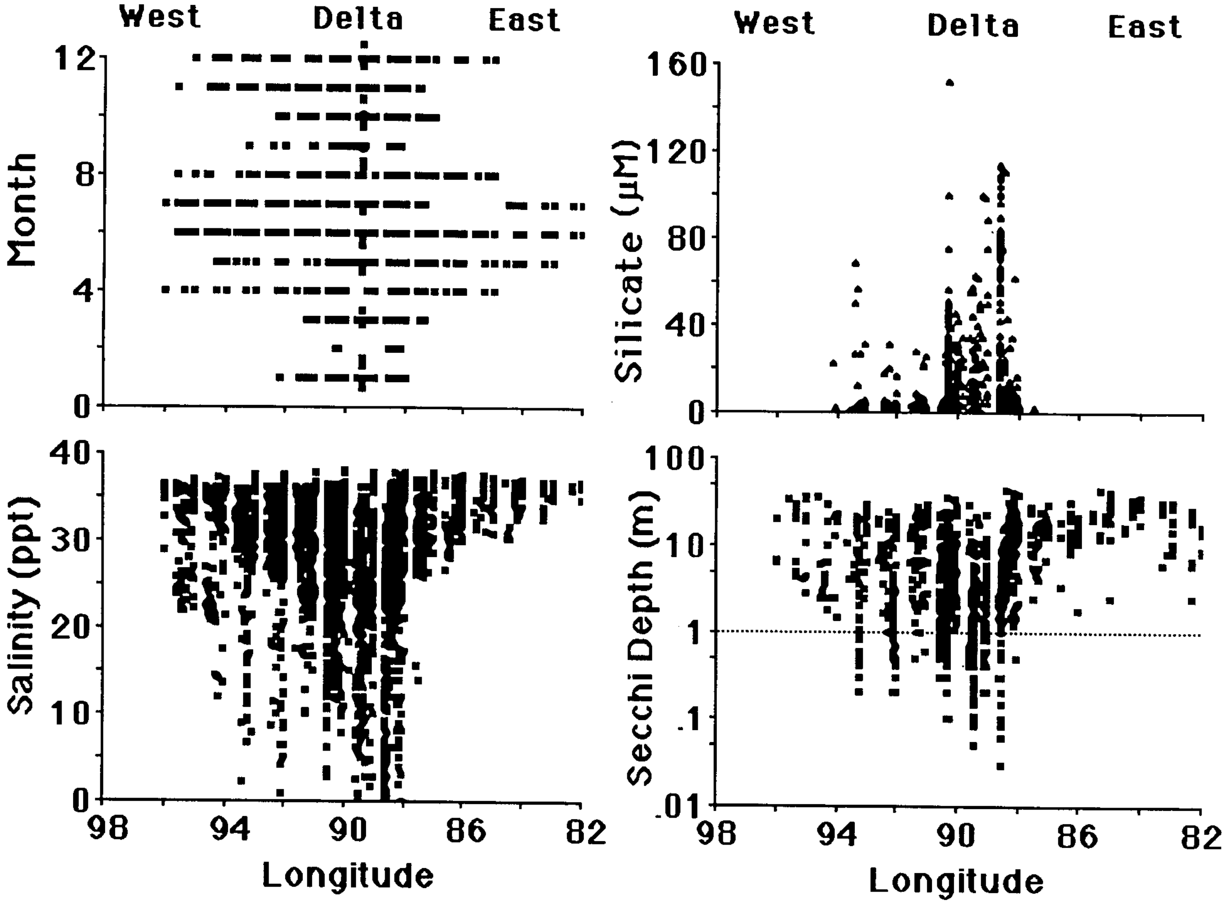


FIGURE 4.3. Plots of surface salinity, Secchi disk depth, and silicate concentrations from a series of hydrographic cruises in 10–100-m water depth for 1972–91, for the months and longitudes indicated. NOTE: The Mississippi River delta is at 90°W; the Atchafalaya River delta is at 92°W. (*From Rabalais et al. 1996.*)

4.2 STRATIFICATION

Freshwater discharge dictates, along with seasonal atmospheric warming, a strong seasonal pycnocline that is necessary for the development and maintenance of hypoxia. The relationships between hypoxia and stratification (either surface-to-bottom differences in sigma t or δ sigma t/depth) within the Louisiana Coastal Current are correlated in time and space (Figure 4.4). This relationship does not always hold, and the depth of the main pycnocline does not always track the depth of the oxycline (Figure 4.5).

The existence of a strong near-surface pycnocline is a necessary condition for the occurrence of hypoxia, while a weaker, seasonal pycnocline guides the morphology of the hypoxic domain (Figures 4.6 and 4.7) (Wiseman et al. 1997). The height above the bottom of the 2 mg/l oxygen isopleth is closely correlated with the height above bottom where the density gradient first achieves a minimum value of 0.01 kg/m^3/m (Figure 4.5). Short-term variability in stratification is due to atmospheric forcing and vertical mixing (Wiseman et al. 1992, 1997); under weak forcing, this variability is largely due to advection (Rabalais et al. 1994).

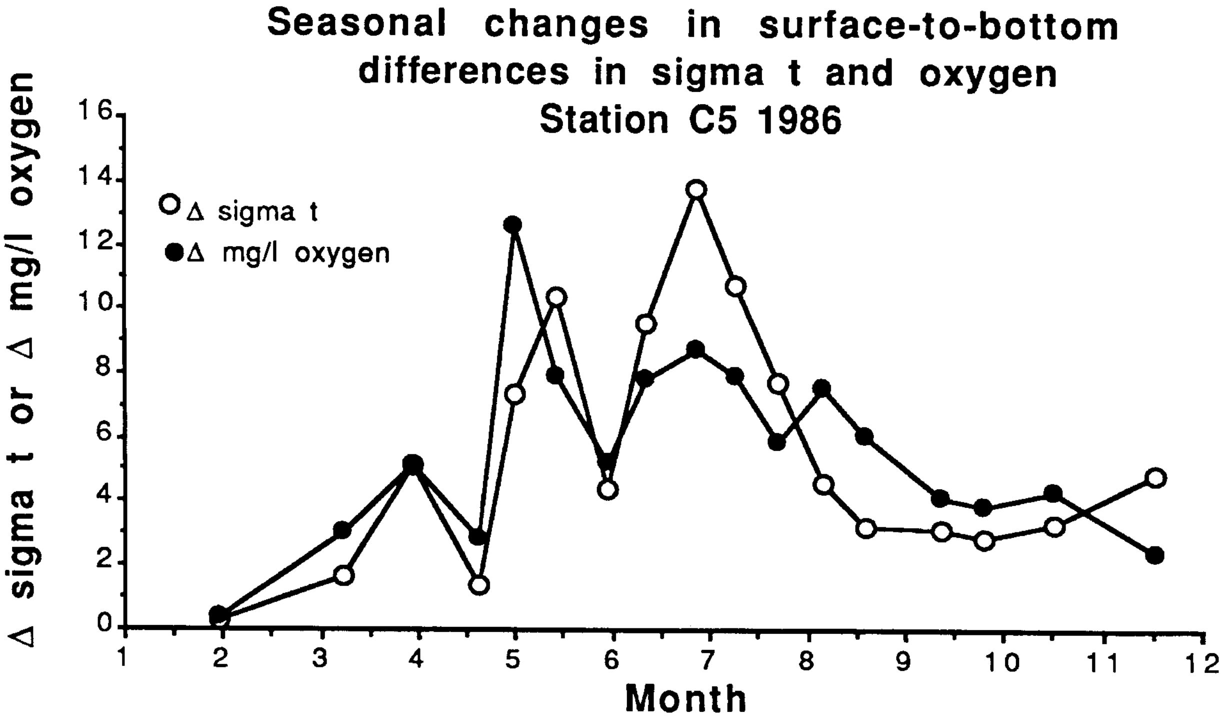


FIGURE 4.4. Differences in surface- to bottom-water values for sigma t and dissolved oxygen concentration by month for station C5 during 1986. (*From Rabalais et al. 1991.*)

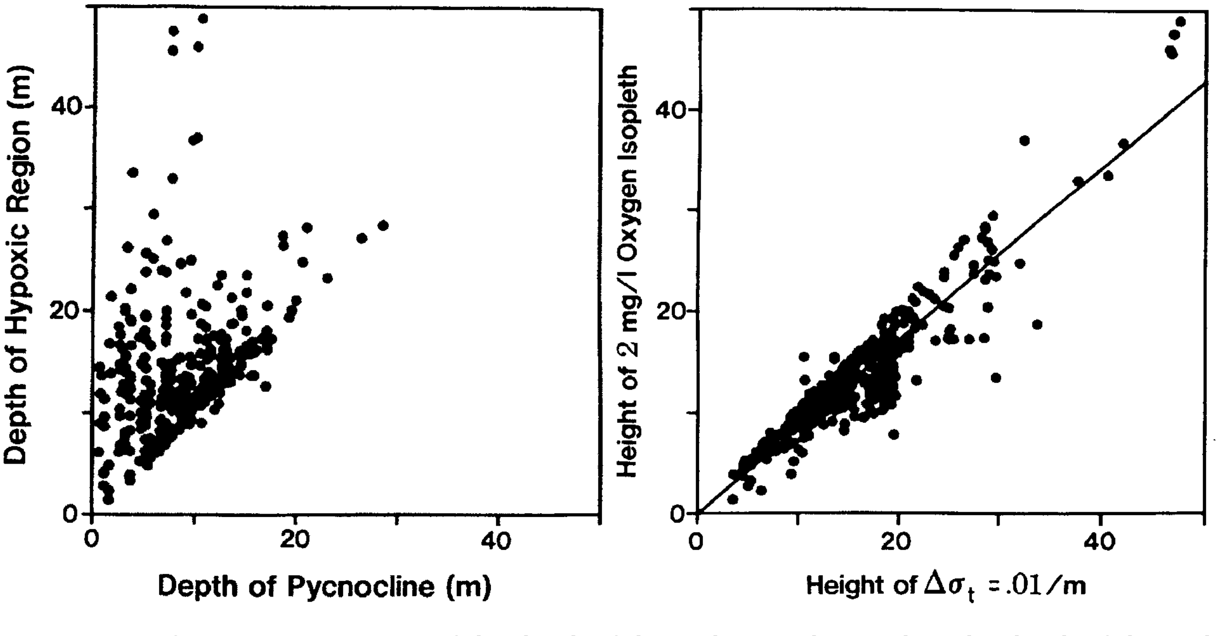


FIGURE 4.5. Left panel: Scatter plot of the depth of the main oxycline against the depth of the main pycnocline for 1985–93 data. Right panel: Scatter plot of the height of the 2 mg/l dissolved oxygen content surface above the bottom against the height of the 0.01 kg/m^3/m density gradient above the bottom. (*From Wiseman et al. 1997.*)

Stratification goes through a well-defined seasonal cycle that generally exhibits maximum stratification during the summer and weakest stratification during the winter. This is due to the strength and phasing of river discharge, wind stirring, regional circulation, and air–sea heat exchange processes. Beginning in early autumn, the summer stratification is disrupted by the first strong cold-air outbreaks. Vertical homogenization of the water column allows reoxygenation of bottom waters, but after the frontal passage, the water column often restabilizes and hypoxic conditions reoccur. These same cold-air outbreaks that cause mechanical stirring of the water by wind stress are accompanied by significant air–sea heat exchanges due to evaporation. The heat loss to the surface waters tends to destabilize the water column. This is partly counterbalanced by fall discharges from the Mississippi and Atchafalaya Rivers. The strongest cold-air outbreaks generally occur during January through March, when river discharge is low or increasing from an earlier minimum. The strong wind and intense heat losses from the surface waters of the coastal ocean result in vigorous stirring of the waters and convective overturning.

In the spring, cold-air outbreaks diminish, river flow is high, and thermal heating begins to stabilize the water column. Stratification develops rapidly, and strong, shore-parallel frontal boundaries associated with the Louisiana Coastal Current develop (Wiseman and Kelly 1994; Murray 1998). During the mid-summer wind reversal along the Texas and Louisiana shelves, strong synoptic weather patterns are infrequent. The wind reversals and associated current reversals, respectively, spread light coastal waters across the shelf due to Ekman coastal divergence and return fresher water initially carried into Texas waters back to the Louisiana shelf. Summer solar heating is intense and contributes to stratification. These processes result in a strong pycnocline that is spread broadly across the shelf. Strongest stratification, however, is associated with the coastal current near shore.

4.3 RELATIVE IMPORTANCE OF PHYSICAL FACTORS

No one factor predicted all elements of the physical structure of the water column during monthly sampling in 1992, but those explaining most of the variability were bottom temperature and salinity, depth, and vertically integrated volume of fresh water (McNaughton 1998). The potential energy anomaly (PEA)is the amount of energy per unit volume that is required to completely homogenize the water column and is a measure of the strength of stratification (Simpson et al. 1993). The strongest relationship among the physical measurements was that of PEA and the thickness of the hypoxic layer, and indicates that as stratification increased, the hypoxic layer thickened (McNaughton 1998). Physical parameters, however, explained less than 40% of the shelf-wide variance of the distribution of dissolved oxygen for the study period of 1992 (McNaughton 1998). Although understanding the physical structure of the water column is relevant to the study of dissolved oxygen cycling, the impact of biological activity on dissolved oxygen concentrations is essential and must be complemented with physical measurements. An integrated and coordinated surveying program is essential for the documentation of oxygen concentrations and the extent of shelf hypoxia and the relevance of physical and biological processes in its formation and maintenance.

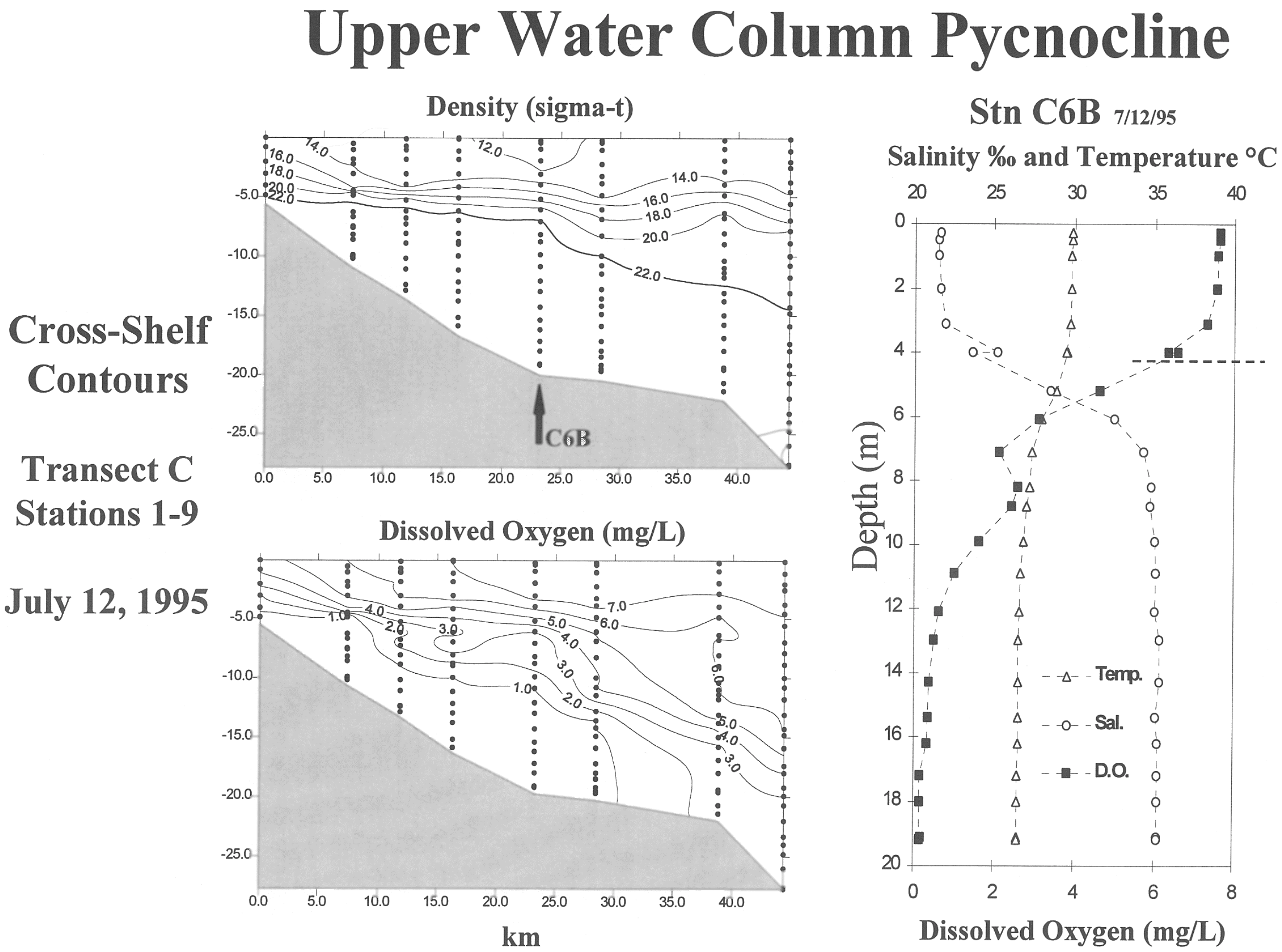


FIGURE 4.6. Cross-shelf contours of density, salinity, and dissolved oxygen for 12 July 1995, with vertical profiles of salinity, temperature, and dissolved oxygen for station C6B. (*Rabalais et al. unpublished data.*)

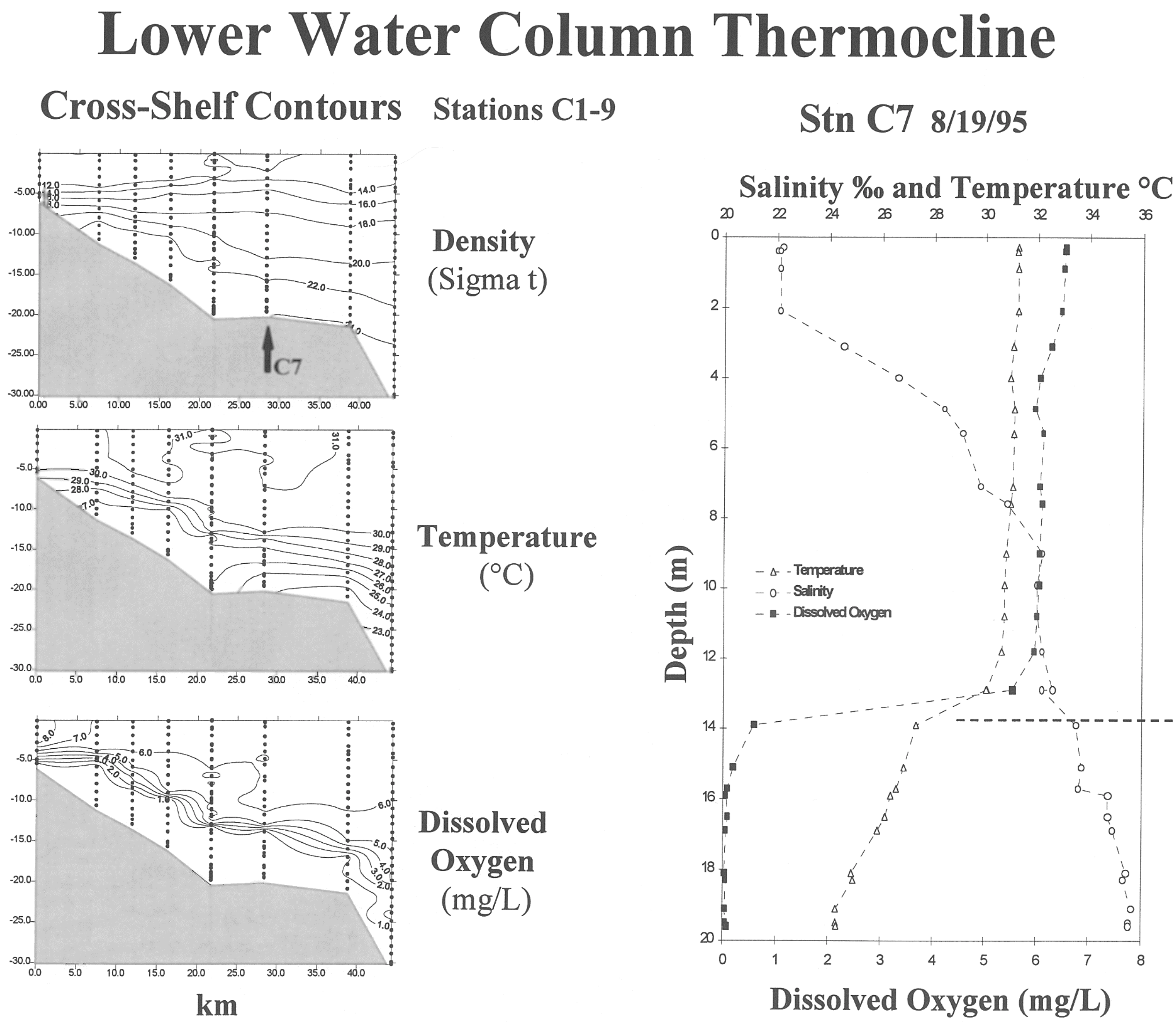


FIGURE 4.7. Cross-shelf contours of density, salinity, and dissolved oxygen for 19 August 1995 with vertical profiles of salinity, temperature, and dissolved oxygen for station C7. (*Rabalais et al. unpublished data.*)

CHAPTER 5

River Discharge and Flux of Materials

5.1 FRESHWATER DISCHARGE

The combined discharges of the Mississippi and Atchafalaya Rivers account for 80% of the total freshwater input to the U.S. Gulf (calculated from U.S. Geological Survey stream-flow data for 37 U.S. streams discharging into the Gulf of Mexico, Dunn 1996). A conservative estimate for the 1930–97 discharge of the Mississippi River system, estimated as the sum of the discharges at Tarbert Landing and Simmsport, is slightly over 19,000 m^3/s (Bratkovich et al. 1994).

Spectra of the discharges exhibit a strong, well-defined annual peak and a broad, flat plateau out to frequencies of about 0.01 cycles per day (cpd). The Atchafalaya and Mississippi River discharges are highly coherent (coherence squared ≥ 0.9) out to frequencies of 0.01 cpd, and the phase between the two signals remains near zero degrees. The long-term peak flow occurs in March, April, and May, and the long-term low flow is in summer and early fall (Figure 3.21). Monthly average flow of the Mississippi River above the diversion point for the Atchafalaya River is a little over 14,000 m^3/s in spring and decreases to below 8,500 m^3/s in August (Rabalais et al. 1991). Although flow is reduced in summer, large-scale circulation patterns facilitate the retention of the fresh water on the shelf (see Section 4.1).

There is significant interannual variability in discharge (Figure 3.21), but the 1820–1992 average rate (multi-decadal time scale) for the Mississippi River at Vicksburg is remarkably stable near 14,000 m^3/s (Figure 5.1) (Turner and Rabalais 1991). Within decades there are upward and downward trends in river discharge; but the long-term average does not differ from zero.

Since the early 1700s humans have altered the morphology and flow of the Mississippi River. Levee construction began around 1717 and increased gradually until the 1880s when the rate was accelerated. Following the 1927 flood, the river levee system was greatly extended and stabilized, and the U.S. Army Corps of Engineers began an extensive flood-control program of channelization and construction of levees and storage reservoirs. As a result, there has been a noticeable decrease in peak discharges since the 1920s and an increase in accumulated water storage in the basin (Everett 1971). Captain Henry Shreve created Old River 80 miles north of Baton Rouge in 1831 to cut off a loop of the Mississippi River to shorten navigation. The upper portion eventually silted up, with the lower limb remaining as a link connecting the Mississippi, Atchafalaya, and Red Rivers. The direction of flow in Old River varied, depending on the height of the Mississippi and Red Rivers. Reversals in flow, however, became less frequent as the Atchafalaya River began enlarging itself through the capture of increasingly greater amounts of the Mississippi's flow. No eastward flow occurred in Old River after 1945. To prevent the Atchafalaya River from becoming the main channel of the Mississippi below Old River, construction was authorized in 1954 for the Old River Control Structure that dammed the Old River and controlled the amounts of water into the Atchafalaya Basin. Flood damage to the control structure in 1972–73 lead to an auxiliary structure that was completed in December 1987. Since 1977 the U.S. Army Corps of Engineers has maintained the amount of water delivered from the Mississippi River to the Atchafalaya Basin at 30%. Some variability occurs in the observed Atchafalaya discharge because of contributions from the Red River to the Atchafalaya.

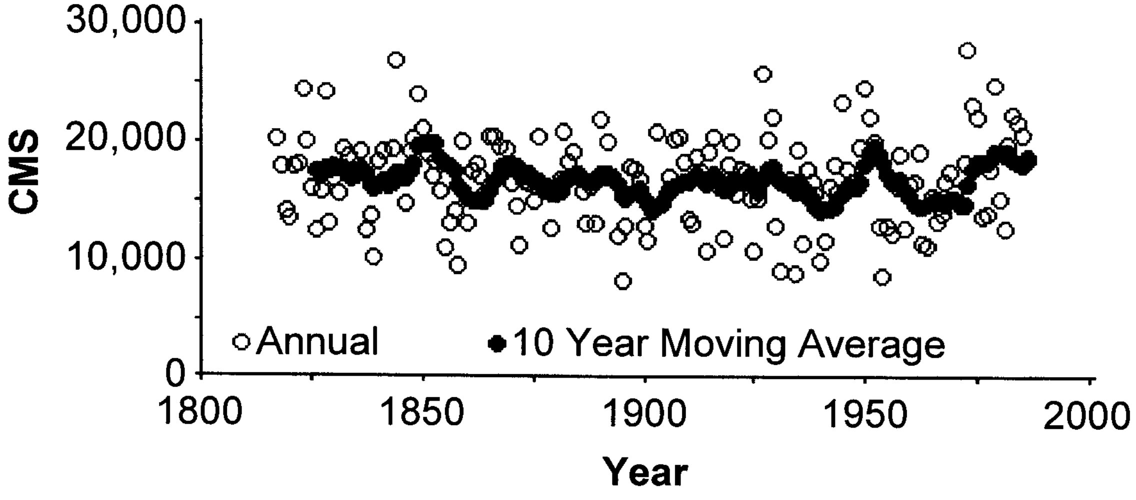


FIGURE 5.1. Annual discharge of the Mississippi River at Vicksburg, Mississippi, with 10-year running average superimposed. (*Data from U.S. Army Corps of Engineers.*)

Bratkovich et al. (1994) reported that the discharge of the Atchafalaya increased during the course of the record examined by them (1900–1992) (Figure 5.2). This trend was statistically significant when tested with a seasonal Kendall tau test for trend (Hirsch et al. 1982) and was associated with the tendency of the river to increasingly capture flow from the Mississippi River proper (up to the 1977 mandate for 30%). An effect likely to occur in the offshore region as a result of increased flow through the Atchafalaya Delta is an increase in stratification west of Atchafalaya Bay and further westward into Texas waters, and a reduction in stratification between the present birdfoot delta and Atchafalaya Bay. Further human-caused or natural capture of more flow down the Atchafalaya would magnify these effects in the offshore area.

Less obvious is a statistically significant and increasing trend in the Mississippi River discharge for 1900–1992 as measured at Tarbert Landing (Bratkovich et al. 1994). It appears to be due to a tendency for increasing discharge in September through December. This period, however, is much less important in the coastal ocean than are spring and summer in the timing of important biological processes that lead to the development of hypoxia or the physical processes important in its maintenance (see Chapters 4 and 6). If a longer period of annual discharge were considered (e.g., Figure 5.1 for the early 1800s to present), the trends since the 1950s are obvious but are concealed within high interannual variability and no long-term change over a century and a half. Whether there are differences or not in long-term trends for freshwater discharge is clearly attributable to the period of record examined. When considering changes to the offshore ecosystem related to physical forcing and/or biological processes stimulated by nutrients, it is important to consider the river discharge for the particular period of concern.

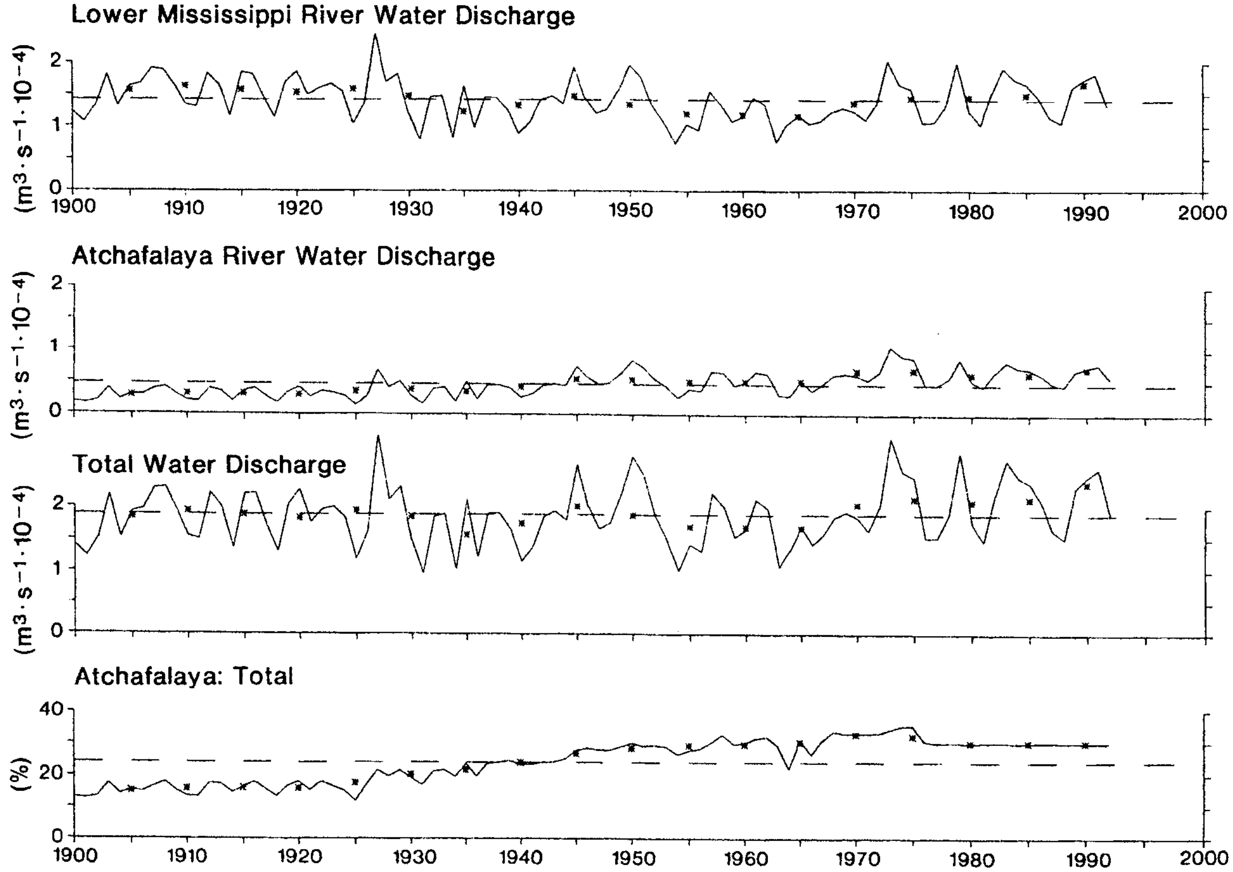


Figure 5.2. The 92-year annual average water discharge time-series data for the lower Mississippi River, Atchafalaya River, and combined flow. NOTE: The lower panel shows the flow ratio for the same time period; Atchafalaya River to total flow. Asterisks are centered, decadal running-mean-averaged values (last values are partly extrapolated). Dashed horizontal lines are 92- year average values. The Atchafalaya River to total flow has been strictly regulated at ~30% since 1977. (*From Bratkovich et al. 1994.*)

5.2 SEDIMENT DISCHARGE

The light regime in the coastal regions has an effect on phytoplankton populations and vice versa and is affected by changes in the quantity, composition, and timing of flux of particulate materials. The suspended sediment loads carried by the Mississippi River to the Gulf of Mexico have decreased by one-half since the Mississippi Valley was first settled by European colonists (Meade et al. 1990). Alterations occurred as a result of deforestation and agriculture, changes in land management, and construction of dams, diversions, and levees. The decrease in suspended sediments has happened mostly since 1950 when the largest natural sources of sediments in the drainage basin were cut off from the Mississippi River mainstem by the construction of large reservoirs on the Missouri and Arkansas Rivers (Meade and Parker 1985). This large decrease in sediments from the western tributaries was counterbalanced somewhat by a five- to tenfold increase in sediment loads in the Ohio River as a result of deforestation and row-crop farming (Keown et al. 1986). The composition of particulate material discharged by the Mississippi River system has also changed during the past century (Kesel 1988).

The high suspended sediment load of the Mississippi River should be an important source of nitrogen to the adjacent Gulf of Mexico (Mayer et al. 1998). Organic nitrogen desorbs from sediment particles in the freshwater/seawater mixing zone and eventually contributes to the total nitrogen load of the river to the Gulf that influences phytoplankton growth. A decrease in sediment load could affect both the particulate and the dissolved organic nitrogen flux. Because Mississippi River suspended sediment load has decreased (Meade 1995) and the dissolved inorganic nitrogen pool has increased from anthropogenic influences (Turner and Rabalais 1991), the relative importance of the organic nitrogen associated with the suspended sediment load may be less for the Mississippi River than for other large world rivers (Mayer et al. 1998). Long-term changes in organic nitrogen flux cannot be documented because that component has been measured only since 1972 (Goolsby et al. 1999). There are indications, however, that organic nitrogen may have contributed to increased production offshore during the 1800s' clearing of land in the Midwest (see Chapter 7).

Because turbidity limits primary production in the plume of the Mississippi River, any reduction in the suspended sediment load could be manifested in clearer water and enhanced productivity. In fact, productivity has increased (see Sections 7.4 and 7.5), but the overall turbidity of the water column has increased in response to increased chlorophyll biomass (Figure 5.3; Turner and Rabalais 1998b). A decade-long change in the light regime is possible for two reasons. First, the suspended load of the Mississippi River has changed with land use, water-control structures, and water-delivery patterns. These changes would be limited to waters < 20 ppt salinity because of the sedimentation of suspended inorganic sediments. The second basic influence on decadal light regime changes involves the doubling or tripling of nitrogen concentration in the Mississippi River since the 1950s–60s (Turner and Rabalais 1991; Justić et al. 1995a; Goolsby et al. 1999), whereas the concentration of silicate declined by 40–50%. These nutrient changes affect the quantities and quality of phytoplankton in the adjacent shelf and, therefore, the pigments. It is noteworthy that the decline in the Secchi disk depth was 3–4% annually, which is about the same rate of increase in nitrate concentration in the Mississippi River (2.5% annually) during the same period (Turner and Rabalais 1991).

5.3 NUTRIENT FLUX

Three nutrients in various forms are important for freshwater and marine phytoplankton growth and production. The pervasive relationships between phosphorus and the biomass and species of freshwater phytoplankton in North America are well established (e.g., Shindler 1977, 1978; Vollenwider and Kerekes 1980). Nitrogen is considered a more dominant influence than phosphorus in estuarine and marine communities (e.g., D'Elia et al. 1986; Harris 1986; Valiela 1984), but not all coastal systems are nitrogen limited (e.g., the Huanghe in China, Turner et al. 1990). Diatoms, an important food group for fish and invertebrates, require silicon to build their cell walls (frustules). Diatoms are the dominant biomass component of many marine and estuarine phytoplankton communities, particularly in the spring. If silica is limited, other non-siliceous forms, such as dinoflagellates or cyanobacteria, may become proportionally more important in the phytoplankton community. Some of these forms are harmful or toxic in bloom proportions. The concentration and relative proportion of these three nutrients to each other are important in phytoplankton production and composition, and subsequent effects on food webs, energy flow, and trophodynamics (Dortch and Whitledge 1992; Justić et al. 1995a, 1995b; Turner et al. 1998).

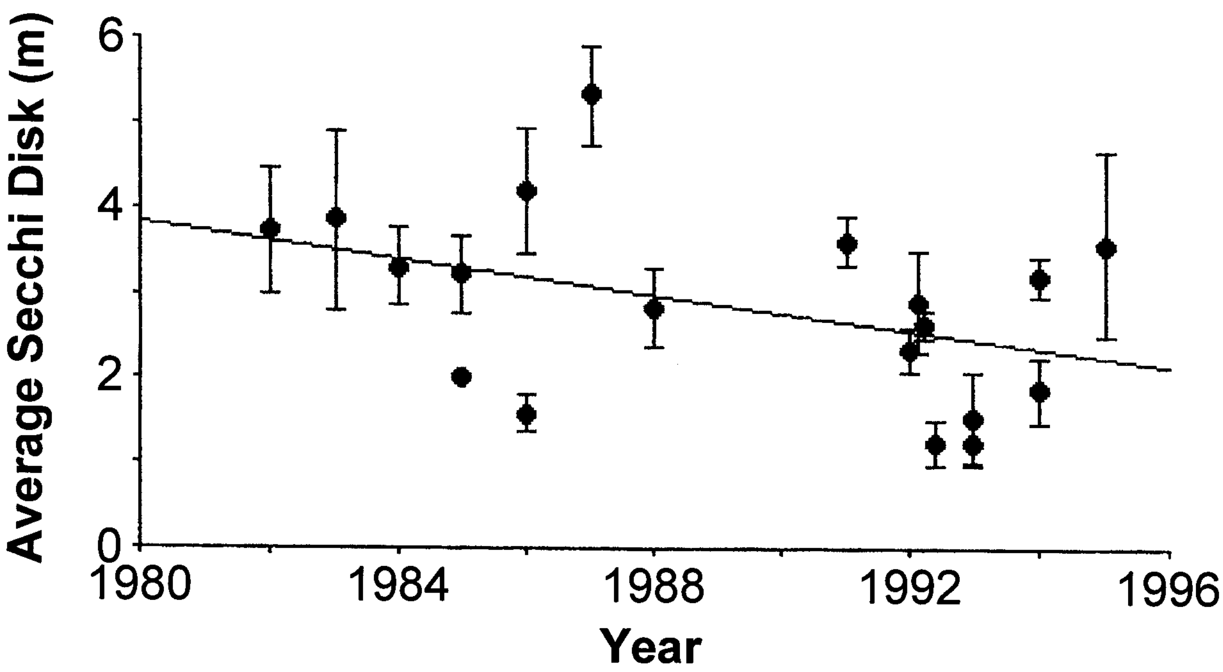


Figure 5.3. Preliminary analysis of the average Secchi disk measurements on the Louisiana shelf west of the Mississippi River Delta. NOTE: The data are restricted to stations with surface-water salinity between 20 and 25 psu and depths between 10 and 100 m. The slope of the regression line is significant at the 8% level of significance. The error bars are ±1 s.e. (*From Turner and Rabalais 1998b.*)

5.3.1 Contribution of Mississippi and Atchafalaya Rivers versus Other Sources

Dunn (1996) calculated the nutrient inflows from 37 U.S. streams discharging into the Gulf of Mexico from Texas through Florida for water years 1972–93. The combined flows of the Mississippi and Atchafalaya Rivers account for 80% of the annual freshwater discharge and 91% of the total nitrogen load. If only streams between Galveston Bay (Texas) and the Mississippi River Delta are considered (i.e., those most likely to influence the zone of hypoxia), the combined flows of the Mississippi and Atchafalaya Rivers account for 96% of the annual freshwater discharge and 98.5% of the total nitrogen load. Similar calculations for annual total phosphorus load are 88% of the total 37 streams and 98% of the streams between Galveston Bay and the Mississippi River Delta for the relative contribution of the Mississippi and Atchafalaya Rivers.

The relative contribution of direct atmospheric deposition of nitrogen to the nitrogen load affecting the area of hypoxia was estimated by R. Artz (see Goolsby et al. 1999) for an area twice the size of the hypoxic zone. The contribution of the Mississippi and Atchafalaya Rivers is 1,500 metric tons/year, and the contribution of atmospheric deposition is 15 metric tons/year, or 1%.

There are no studies of ground-water discharge to coastal waters for the coastlines of Texas, Louisiana, Mississippi, or Alabama (J. Cable, personal communication). Studies do exist from Florida and the Yucatan Peninsula, but these environs differ substantially in geologic formation (karsts). Ground-water sources to the area affected by hypoxia are unlikely to be important because of the lack of shallow aquifers along the Louisiana coast. Any contributions of ground water to the coastal waters of the northern Gulf would probably become entrained in the Louisiana Coastal Current, where strong, shore-parallel frontal boundaries develop and where net flow is generally to the west alongshore, except for reversals in mid-summer. For comparisons with other sources—e.g., riverine and atmospheric—it may be worthwhile to provide some information from other areas.

For the northeastern Gulf of Mexico, the total N input from ground water to Apalachee Bay was estimated to be 11 kg/ha/yr (J. Cable, personal communication), which slightly exceeded the combined flux of total nitrogen from the Ochlockonee and Sopchoppy Rivers (calculated from data in Fu and Winchester 1994). Ground-water inputs to the South Atlantic Bight were estimated (based on summer conditions) to be about 40% of the river-water flux to the coastal waters (Moore 1996). These estimates demonstrate the potential importance of ground-water flux to coastal waters. On the scale of contributions from the Mississippi and Atchafalaya Rivers, however, estimates for the northeastern Gulf of Mexico (11 kg/ha/yr) would be a minimal contribution (if similar values were to be expected from the central and northwestern Gulf), compared to the Mississippi and Atchafalaya River flux. The contribution of ground water as a nutrient source fueling hypoxia is unknown but unlikely, and the potential for transfer in a cross-shelf direction is minimal.

The relative contribution of offshore sources of nutrients from upwelled waters of the continental slope is unknown. Onwelling of nitrate from deeper waters may be important in shelf-edge (100-m depth range) cycling of carbon and nitrogen (Walsh 1988, 1991). Occasionally, mixing diagrams of riverine nutrients with the saline, nutrient-poor Gulf of Mexico waters are nonlinear in a way that implies another source of nutrients. The source has been suggested to be offshore nutrients from deeper-water intrusion (Lopez–Veneroni and Cifuentes 1994; Lorenz et al. in press). Preliminary studies of total dissolved nitrogen and particulate nitrogen using naturally occurring nitrogen isotopes are under way (L. Cifuentes, personal communication), and the relative contribution is unknown.

The Mississippi and Atchafalaya Rivers contribute, by far, the major sources of fresh water and nutrients from stream flow to the northern Gulf of Mexico. Direct atmospheric deposition is minimal, ground-water contributions are unlikely, and the contribution of upwelled nutrients is unknown. The best current knowledge is that the outflows of the two rivers dominate the nutrient loads to the continental shelf where hypoxia is likely to develop.

5.3.2 Flux and Concentration of Nutrients

Several researchers have documented changes in Mississippi River nutrient concentration and flux over varying periods (Smith et al. 1987; Turner and Rabalais 1991, 1994a, 1994b; Justić et al. 1995a, 1995b; Rabalais et al. 1996; Meade 1995; see CENR Topic 3 report by Goolsby et al. 1999). Differences in results from these studies come from analysis of different time periods, different analytical methods, and different calculations of flux. There are also differences, depending on whether the concentration of a constituent in the lower Mississippi as it approaches the Gulf or its flux to the Gulf is under consideration. Results generally concur that Mississippi River nutrient concentrations and loading to the adjacent continental shelf have changed dramatically this century, with an acceleration of these changes in the last four decades, depending on the constituent of concern.

Turner and Rabalais (1991) and Justić et al. (1995b) examined water quality data for the lower Mississippi River for dissolved inorganic nitrogen (as nitrate), phosphorus (as total phosphorus), and silicon (as silicate). The mean annual concentration of nitrate was approximately the same in 1905–6 and 1933–34 as in the 1950s, but has doubled (or tripled, depending on the comparative periods) from the 1950s and 1960s values (Figures 5.4 and 5.5 and Table 5.1). The flux of nitrogen has also increased over the last three decades (Figure 5.6). These results were corroborated by Goolsby et al. (1999), who documented that the average mean nitrate concentration at St. Francisville during 1980–96 was more than double the average concentration during 1955–70. The mean annual nitrogen flux (determined by Goolsby et al. 1999 from discharge weighted regression models) approximately tripled in the last 30 years, with most of the increase occurring between 1970 and 1983. The mean annual N flux has changed little since the early 1980s, but there are large year-to-year variations in N flux because of variations in precipitation. The increase in total nitrogen is almost entirely due to changes in nitrate concentration.

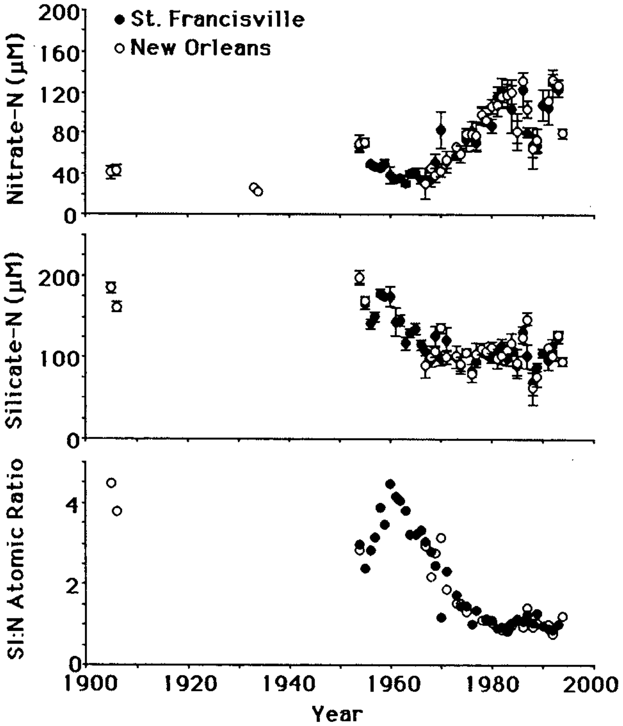


FIGURE 5.4. Changes in nitrate and silicate and the Si:N ratio from 1905 through 1994. NOTE: ● for St. Francisville, ❍ for New Orleans; bars are ± s.e. of the average of monthly means. (*Modified from Turner and Rabalais 1991 in Rabalais et al. 1996.*)

The mean annual concentration of silicate in the lower Mississippi River was approximately the same in 1905–6 as in the early 1950s; then it declined by 50% (Turner and Rabalais 1991). Justić et al. (1995b) documented a 30% decrease in silicate concentrations between the periods 1960–62 and 1981–87. Goolsby et al. (1999) noted a 40% decline in silicate concentrations from the 1950s to the mid-1970s when they stabilized. They further noted that there was no long-term trend in silicate flux as observed for nitrate. Concentrations of nitrate and silicate appear to have stabilized, but trends are masked by increased variability in the 1980s and 1990s data.

There are no substantial records of total phosphorus concentrations in the lower Mississippi River before 1973, and subsequent values vary greatly among years. Goolsby et al. (1999) found no long-term trend in orthophosphate or total P from 1973 to 1996. Turner and Rabalais (1991) noted a trend for an increase in orthophosphate but stated that extreme variability in the data made it unlikely that a statistical trend could be identified. Justić et al. (1995b) applied a linear least-squares regression on the 1973–87 total P data and estimated ($p < 0.01$) that the total P concentration increased twofold between 1960–62 and 1981–87 (Table 5.1). This result was corroborated with current measurements and historical data from coastal waters adjacent to the Mississippi River discharge (Thomas and Simmons 1960).

Thus, there is agreement that nitrate concentration and flux have either doubled or tripled since the turn of the century and the 1950s-60s, and that levels have plateaued but exhibit variability from year to year. The increase in total N is due to the increase in the nitrate component. There is also agreement that the concentration of silicate has decreased, anywhere from 30% to 50%, depending on the period of record. The phosphorus data are of shorter duration and less robust for determining long-term trends, for which there is disagreement of “no trend” to “increase of twofold” from the 1960s to the 1980s, based on a regression of the 1973–87 total P data and parallel data from the coastal system.

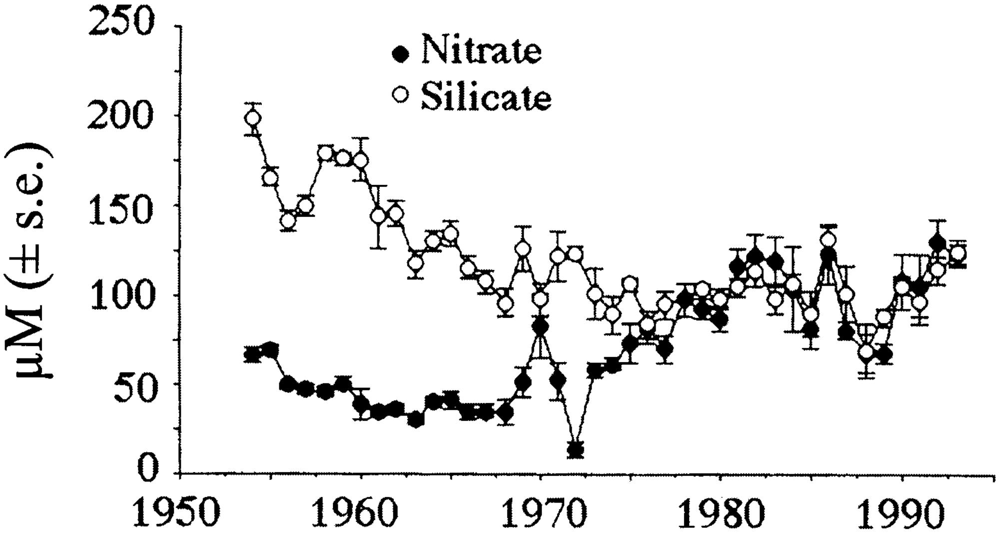


Figure 5.5. Average annual concentration (± s.e.) of nitrate and silicate in the Mississippi River at New Orleans. (*From Turner et al. 1998.*)

TABLE 5.1. Historical changes in concentrations and atomic ratios of nitrogen, phosphorus and silica in the lower Mississippi River and the northern Gulf of Mexico. Note: $p < 0.001$—highly significant difference in nutrient concentrations between the two periods, based on a two-sample t-test. (*From Rabalais et al. 1996, and modified from Justić et al. 1995b.*)

Nutrient Concentrations and Average Atomic Ratios		Mississippi River		Northern Gulf of Mexico	
		1960–62[4]	*1981–87*	*1960*[5]	*1981–87*
Nutrient Concentrations (µM)					
Nitrogen[1]	Mean	36.5	114	2.23	8.13
	No. of Data	72	200	219	219
	Standard Error	2.9	6.0 ($p < 0.001$)	0.16	0.60
Phosphorus[2]	Mean	3.9	7.7	0.14	0.34
	No. of Data	-	234	231	231
	Standard Error	-	0.4 ($p < 0.001$)	0.01	0.02
Silica[3]	Mean	155.1	108	8.97	5.34
	No. of Data	72	71	235	235
	Standard Error	7.5	4.3 ($p < 0.001$)	0.55	0.33
Average Atomic Ratios					
Silica:Nitrogen		4.2	0.9	4.0	0.7
Nitrogen:Phosphorus		9	15	16	24
Silica:Phosphorus		39.8	14	64	16

[1] N-NO$_3$ for the Mississippi River, dissolved inorganic nitrogen (DIN = NO$_3$ + NH$_4$ + NO$_2$) for the northern Gulf.
[2] Total P for the Mississippi River, reactive P for the northern Gulf of Mexico.
[3] Reactive Si.
[4] Turner and Rabalais 1991 for N and Si, reconstructed for P.
[5] Reconstructed data.

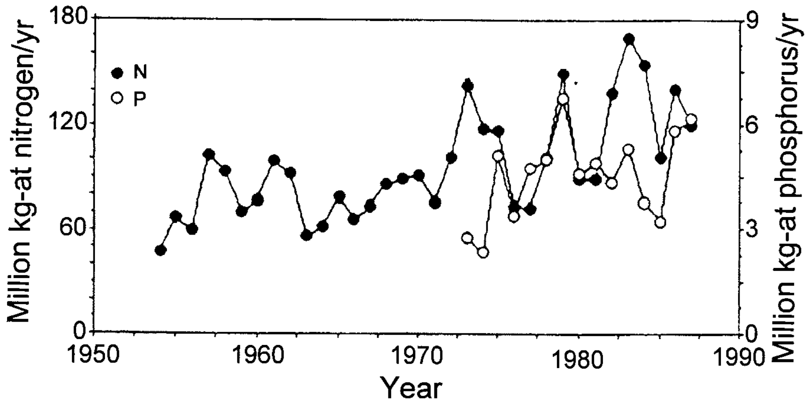


Figure 5.6. Nitrogen and phosphorus loading from the Mississippi and Atchafalaya Rivers to the Gulf of Mexico from 1954 to 1987. (*From Turner and Rabalais 1991.*)

Figure 5.7 shows the relationship between the concentration of nitrate and total nitrogen at St. Francisville. Both the average nitrate and the total nitrogen concentrations have increased in recent decades (Turner and Rabalais 1991), but the percent nitrogen as nitrate has varied between 30% and 60%. This percentage leveled off in the l980s–90s at 60%. From 1977 to 1994, the percent of the total nitrogen pool averaged 59% nitrate and 37% organic nitrogen. The remaining 4% was ammonium (3%), nitrite (1%), and unidentified dissolved organic nitrogen molecules. The importance of dissolved forms other than nitrate and the dissolved organic and particulate organic nitrogen components is not being dismissed, but less is known about their relative contribution to the total flux over a longer time frame and the related biological processes on the adjacent shelf. Studies from the Mississippi River plume support the view of an active microbial population capable of utilization of organic nitrogen and rapid rates of regeneration of inorganic nutrients (Gardner et al. 1994, 1997; Pakulski et al. 1995; Bode and Dortch 1996).

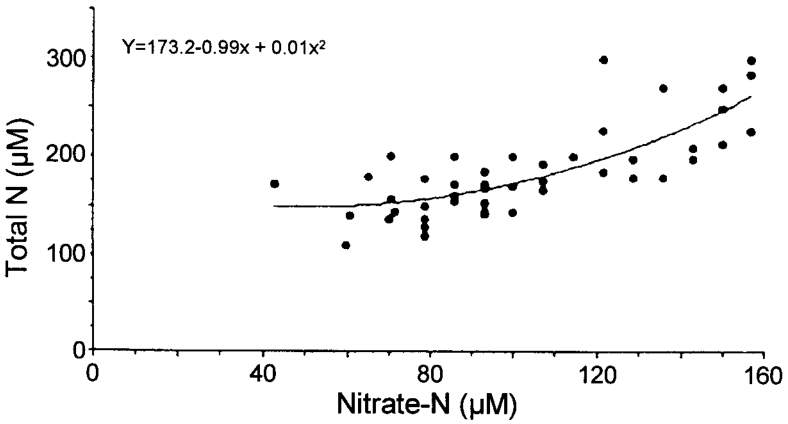


FIGURE 5.7. Relationship between the concentration of nitrate–nitrogen and total nitrogen in the lower Mississippi River at St. Francisville, Louisiana. (*Turner and Rabalais 1991.)*

The effluent of the Atchafalaya River at Morgan City (0.5 km from the terminus of the river) varies somewhat from the Mississippi River mainstem (Turner and Rabalais 1991). The concentrations of nitrate, silicate, and total phosphorus at Morgan City (Atchafalaya River) were 69%, 94%, and 130%, respectively, of that in the Mississippi River at St. Francisville for the combined data for the years 1973–87. Differences between nutrient concentrations of the two discharges are presumably a consequence of differences in the water quality of the Red River (which mixes with the Mississippi to form the Atchafalaya), compared with that of the Mississippi, inputs within the Atchafalaya Basin, processing and transformations within the basin, or combinations of these.

5.3.3 Seasonality

The seasonal patterns in nitrate and silicate concentrations have also changed during this century (Figure 5.8) (see also Goolsby et al. 1999, Topic 3 report). There was no pronounced peak in nitrate concentration prior to 1960, whereas there was a spring peak from 1975 to 1985. Because there was no seasonal peak in nitrate prior to the 1960s, nitrogen flux closely paralleled river discharge. Currently, a similar pattern (flux is related to discharge) holds (Alexander et al. 1996; Goolsby et al. 1999), but the flux of nitrogen per volume discharge is greater than historically. The relationship of nutrient flux with discharge is not perfect, however, because there is some seasonality of nitrate flux independent of river discharge (Justić et al. 1997) (see Section 8.3). In contrast, a historical seasonal summer–fall maximum in silicate concentration is no longer evident. Currently, there is nearly a twofold difference in nitrate supply over the course of the year (Turner and Rabalais 1991), but only small annual variations in the silicate and total phosphorus supply (Figure 5.9).

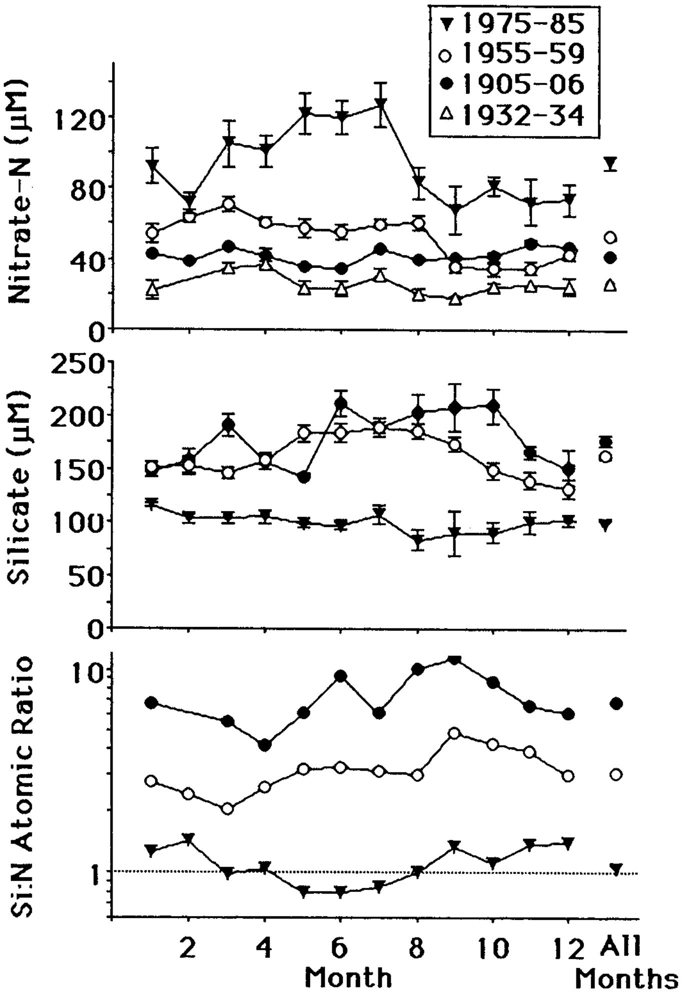


FIGURE 5.8. Monthly average concentrations of nitrate and silicate, and Si:N ratio in the lower Mississippi River for the periods indicated. (*Modified and updated from Turner and Rabalais 1991, 1994a; from Rabalais et al. 1996.*)

5.3.4 Nutrient Ratios

The proportions of dissolved Si, N, and P in the lower Mississippi River have changed historically such that they now closely approximate the Redfield ratio (Si:N:P = 16:16:1; Figures 5.4, 5.5, 5.8, and 5.10) (Justić et al. 1995a, 1995b). The Si:N atomic ratio was approximately 4:1 at the beginning of this century, dropped to 3:1 in 1950, and then rose to approximately 4.5:1 during the next 10 years, before plummeting to 1:1 in the 1980s. The ratio appears stable at 1:1 through 1997 with little variation (Figures 5.4 and 5.5). The Si:P ratio decreased from 40 to 14, and the N:P ratio increased from 9 to 15.

By applying the Redfield ratio as a criterion for stoichiometric nutrient balance, one can distinguish between P-deficient, N-deficient, and Si-deficient rivers and those having a well-balanced nutrient composition. The annual nutrient ratios for the Mississippi (1981–87 database) show an almost perfect coincidence with the Redfield ratio and suggest a balanced nutrient composition (Figure 5.10). With nutrient concentrations so closely balanced, Justić et al. (1995b) proposed that any nutrient can become limiting, perhaps in response to small differences in nutrient supply ratios.

The water quality changes in the Mississippi River are not unique among world rivers (Justić et al. 1995a; Howarth et al. 1996). The Mississippi is one of several rivers in which the concentrations and proportions of nitrogen, phosphorus, and silicate have changed over many decades as a result of anthropogenic activity in the watershed. Justić et al. (1995a) examined nutrient data from 10 large world rivers. Pristine rivers (e.g., the Yukon, Mackenzie, Amazon, and Zaire) deliver silicate to the coastal ocean in great stoichiometric excess over nitrogen and phosphorus, relative to the Redfield ratio (Si:N:P = 16:16:1), and nutrient requirements of diatoms (Figure 5.10). Consequently, they tend to have nitrogen- or phosphorus-deficient plumes and support nitrogen or phosphorus limitation in the coastal waters.

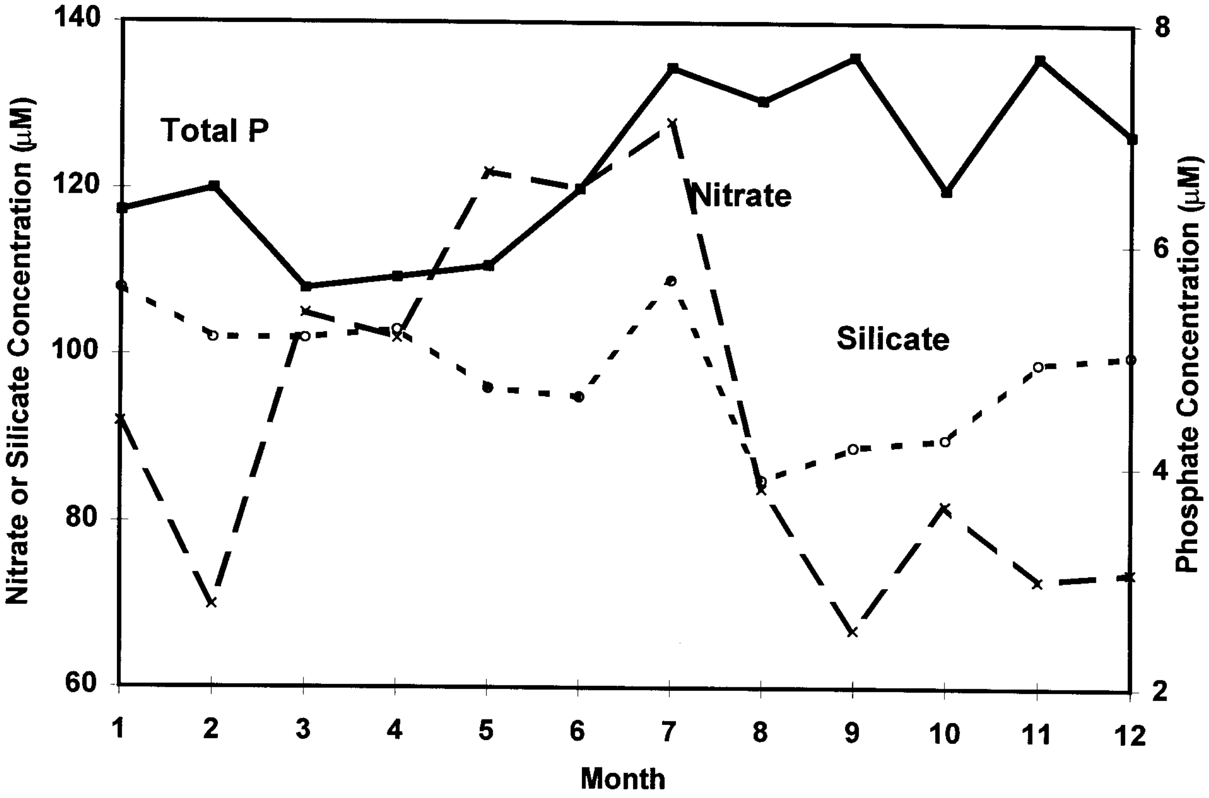


FIGURE 5.9. Monthly variation in average (1975–85) nitrate, silicate, and total phosphorus concentrations in the Mississippi River at St. Francisville, Louisiana. (*Redrawn from Figure 4, Turner and Rabalais 1991.*)

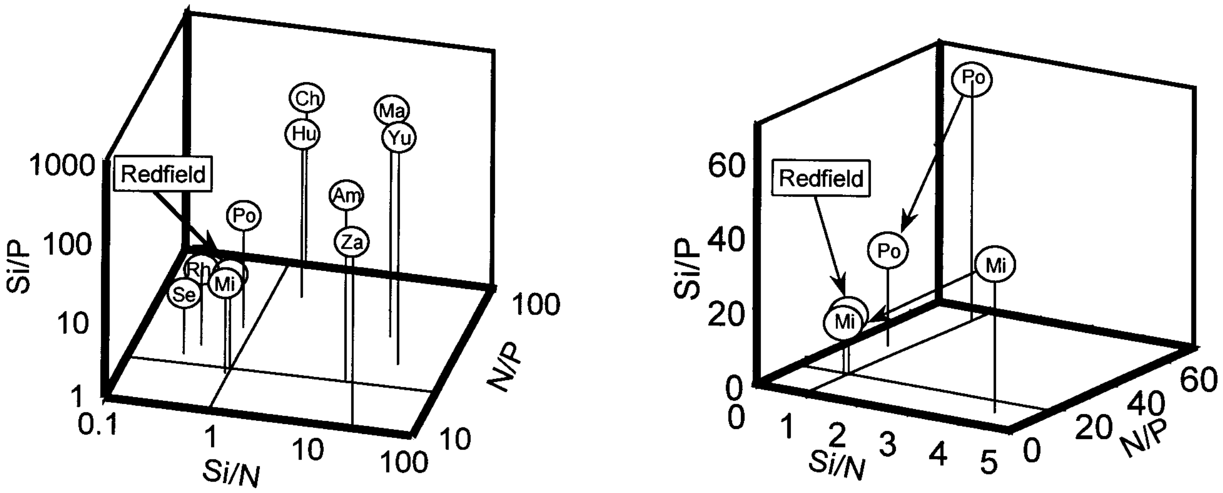


FIGURE 5.10. Left panel: Clustering of mean atomic ratios of dissolved inorganic nitrogen (N), phosphorus (P), and silica (Si) in the Amazon, Changjiang, Huanghe, Mackenzie, Mississippi, Po, Rhine, Seine, Yukon, and Zaire Rivers. Right panel: Changes in the atomic ratios of N, P, and Si between 1960–62 and 1981–87 in the Mississippi River and between 1968–70 and 1981–84 in the Po River. NOTE: The Redfield ratio, Si:N:P = 16:16:1 in both diagrams. (*Modified from Justić et al. 1995b.*)

In anthropogenically affected rivers—such as the Po, Mississippi, Rhine, and Seine—proportions of the nutrients have changed in such a way that they now approximate the Redfield ratio. Thus, in the coastal waters affected by these rivers, the silicate excess has diminished, and a more balanced nutrient structure has resulted. Expected consequences would be an increase in surface primary productivity and possibly increased frequency of hypoxia. These effects have been clearly demonstrated in the northern Adriatic Sea adjacent to the Po River (Justić 1991a) and also in the Mississippi River (Chapters 7 and 8). Rivers with increased nitrogen and phosphorus in relation to silicate may affect a limitation to the productivity of diatoms while increasing productivity. The result would be a shift in dominance from diatoms to nonsiliceous taxa. Proportions of nutrients in some European rivers, e.g., the Rhine and Seine, already indicate silicate deficiency with respect to the nutrient requirements of diatoms. The increasing frequency of nondiatom blooms in the coastal waters of northwestern Europe (Smayda 1990; Hallegraeff 1993) is consistent with the progression toward silicate deficiency.

Although the concentrations of nitrate-N and silicate currently average near 100 μM on an annual basis, there is much variability about this average, but a remarkable coherence since the early 1980s (Figure 5.5) near a ratio of 1:1. Total P averages 7.4 μM at the mouth of the Mississippi River and is lowest in the spring. Thus, the nutrient supply ratios vary around the Redfield ratios on a seasonal basis, with silicate and phosphorus in the shortest supply during the spring, and nitrogen more likely to be limiting (based on ratios) during the rest of the year (Figure 5.11).

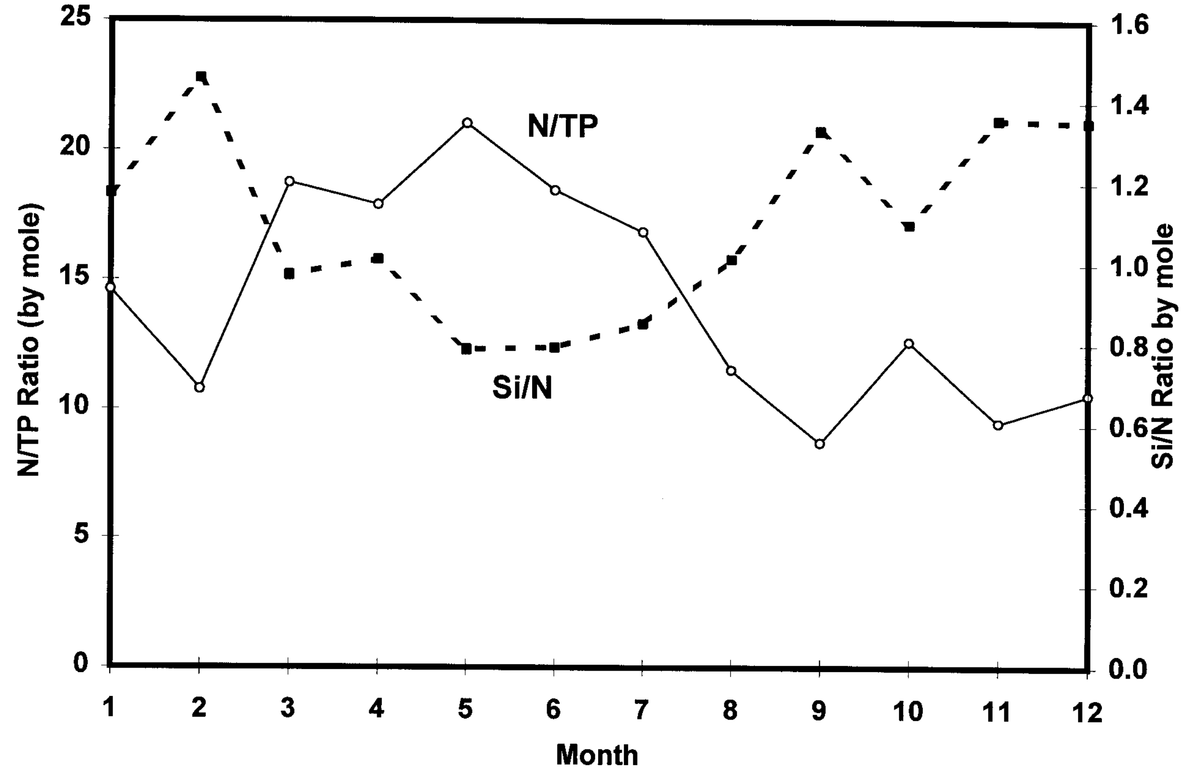


FIGURE 5.11. Monthly variation in average (1975–85) nutrient ratios in the Mississippi River at St. Francisville, Louisiana. (*Data from Figure 4, Turner and Rabalais 1991.*)

5.3.5 Anthropogenic Influences

Over recent decades and the last century and a half, the landscape of the Mississippi River and human activities in it have changed dramatically, with implications for changes in nutrient concentration, flux and relative proportions, coastal ecosystem change (nutrient-enhanced productivity), and worsening oxygen stress. Major alterations in the morphology of the main river channel, delivery of water (see Section 5.1), and widespread landscape alterations and land-use patterns in the watershed have altered the natural processing of nutrients (Goolsby et al. 1999).

Better than half of the original wetlands in the United States have been lost to drainage practices (Zucker and Brown 1998). Much of this wetland loss is related to agricultural production in areas that were swampy and too wet to farm. Within the Mississippi River Valley, 56% of the wetlands have been lost to agriculture, navigation, reservoirs, and leveeing (Winger 1986).

Drainage systems may have a positive impact on some nonpoint-source pollution problems in comparison to agricultural land without drainage. For example, under certain conditions, artificial drainage lowers soil erosion by increasing the movement of water through the soil profile and thus reducing runoff. The reduction in loss of phosphorus ranges up to 45%. Subsurface drainage, however, expedites the transport of nitrate-N from the soil zone to surface waters. This management practice, coupled with the increase in fertilizer applications (see below), can only increase the flux of nitrate-N from agricultural fields to the receiving waters of the Mississippi River watershed.

In addition to landscape changes, anthropogenic inputs of nitrogen and phosphorus have increased from agriculture, point sources, and atmospheric deposition. The estimate of current nitrogen export from the Mississippi River watershed over “pristine” river (pre-agricultural and pre-industrial condition) nitrogen export is a 2.5- to 7.4-fold increase (Howarth et al. 1996). Fertilizer accounts for well over two-thirds of the estimated increase in their analysis, and atmospheric deposition, the remainder. According to Goolsby et al. (1999), agricultural activities are the lar-gest contributor of nitrate to streams, and of all the major nitrogen inputs to croplands, only fertilizer and legumes have increased since the 1950s.

Data on the nitrogen and phosphorus fertilizer use and form in the United States and the world are available in annual reports of the United Nations (year = July through June) and the U.S. Department of Agriculture (year = January through December). U.S. nitrogen and phosphorus fertilizer use began in the mid-1930s and climbed to a peak around 1980 (Figure 5.12) Nitrogen fertilizer use in the United States appears to have reached a plateau beginning in the 1980s, whereas phosphorus fertilizer use stabilized in 1980 and then dropped slightly.

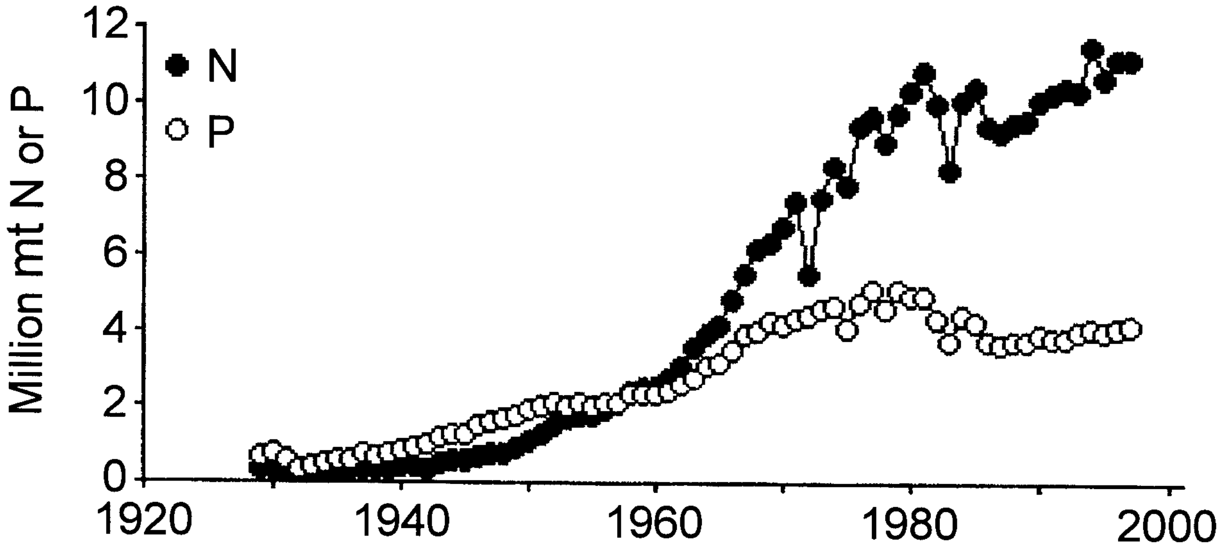


FIGURE 5.12. Nitrogen (as N) and phosphorus (as P_2O_5 equivalent) fertilizer use this century in the United States up to 1996–97. (*Modified from Turner and Rabalais 1991.*)

The form of nitrogen and phosphorus fertilizers used has changed during the last several decades (Turner and Rabalais 1991). In 1929, nitrogen fertilizer was 48% ammonia, 19% nitrate, and the balance mostly organic nitrogen. By 1949, the proportion was 79.5% ammonia and 12% nitrate, and the percent ammonia has remained the same or increased slightly since then. During 1981–85, 42% of the nitrogen fertilizer and 37% of the phosphorus fertilizer used annually in the U.S. was applied in states that are partly or completely in the Mississippi River watershed, where it equaled 4.2 million mt of nitrogen (as N) and 0.53 million mt of phosphorus (as P).

The rise in nitrate concentrations in the lower Mississippi River since 1960 was strongly related (R^2 = 0.74) to increased nitrogen fertilizer application in the watershed (Turner and Rabalais 1991) (Figure 5.13). The change in silicate since 1960 is strongly and inversely correlated with phosphorus fertilizer use ($R^2 = 0.79$). The decrease appears to be a consequence of upstream phosphorus additions that stimulated freshwater diatom production and an eventual burial in freshwater sediments of silica in diatom remains (e.g., Schelske and Stoermer 1971; Schelske et al. 1986), thus reducing the annual supply of riverine silicate to coastal waters. Fertilizer use reached a plateau in the late 1980s. At the same time, both the rise in river nitrate concentration and the fall in silicate concentration stopped.

Decreases in suspended sediments or damming of the distributaries within the Mississippi River system have been proposed as an alternative explanation for the observed decrease in silicate concentration in the lower Mississippi River. Silicon may be lost from the water column because it becomes attached to sediments or deposited in the sediments that collect behind dams or in reservoirs and is subsequently not recycled to the water column. This explanation seems unacceptable because (1) suspended sediment concentrations stabilized in the mid-1950s (Kesel 1988), whereas the silicate concentrations continued to decline; (2) silicate concentrations changed coincidentally with phosphorus fertilizer use; and (3) the silicon-depletion hypothesis has proven an acceptable interpretation of water quality changes in lakes (e.g., Schelske et al. 1983, 1986).

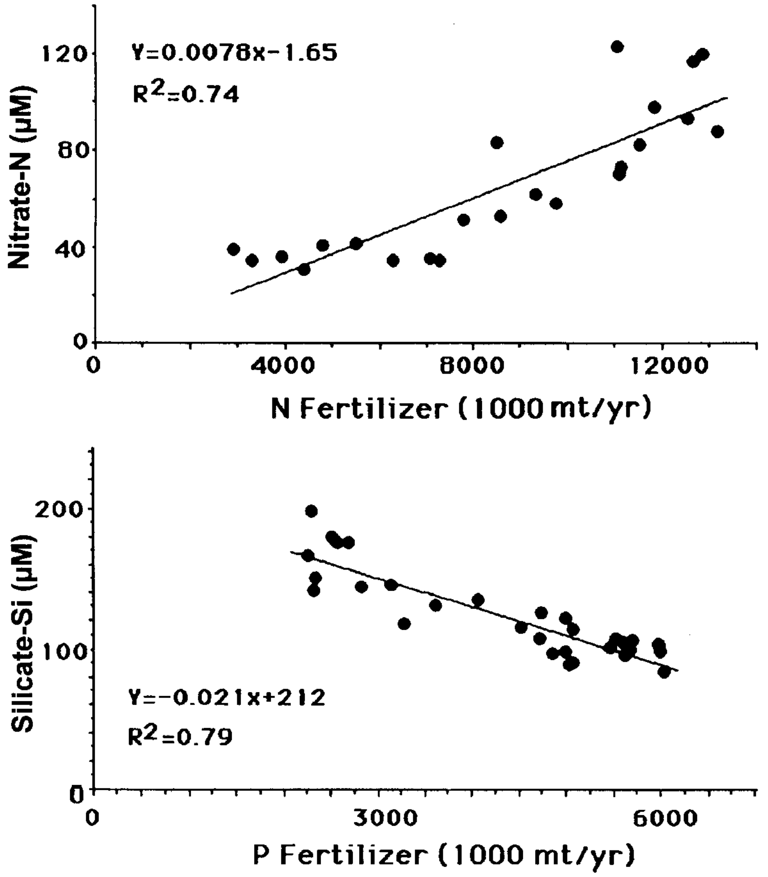


FIGURE 5.13. Relationship between fertilizer use and water quality at St. Francisville, Louisiana. Top panel: Nitrogen (as N) fertilizer use in the United States and average annual nitrate concentrations from 1960 to 1985. Bottom panel: Phosphorus (as P_2O_5) fertilizer use in the United States and average annual silicate concentrations from 1950 to 1987. NOTE: Fertilizer use and water quality for the same years are compared, because water retention on the watershed is less than one year. (*From Turner and Rabalais 1991.*)

The longer retention time in freshwater areas, however, may exacerbate the silicon depletion scenario due to nutrient enrichment. The slow decrease in silica concentration in the Mississippi River is consistent with a biological mechanism, as a result of P enrichment, increased water residence time, or both. Loss in silicate delivery to the northwestern Black Sea, with subsequent eutrophication and harmful algal blooms, was attributed to damming on the Danube River and sequestering of silicate on sediments (Humborg et al. 1997). These authors, however, failed to acknowledge the known acceleration of both phosphorus and nitrogen loads in the Danube. Also, their “step” reduction in silicate concentration attributed to the building of the Iron Gates is in reality a gradual decline similar to that seen in the Mississippi River silicate concentration (see Section 5.3.2). Also, there are several hypotheses concerning changing Si:N ratios within the watershed related to changes in land-use patterns, cropping systems, and types of crops.

5.4 CHANGES IN OFFSHORE NUTRIENTS

Justićet al. (1994, 1995a, 1995b) analyzed extensive nutrient data sets from the northern Gulf of Mexico to examine how the coastal nutrient structure may reflect long-term changes in the proportions of dissolved Si, N, and P in riverine loads. Because fully reliable long-term data sets to examine the nutrient composition 30 years ago were not available, the authors reconstructed the past coastal nutrient composition for the last 30 years by assuming that the relative proportion of nutrients in the river-dominated coastal waters reflects changing composition of riverine nutrients (Table 3.1, Justić et al. 1995a). This assumption was made in view of the fact that the Mississippi River is the most important source of nutrients to the northern Gulf of Mexico.

Comparison of the reconstructed data with the available historical nutrient data (Thomas and Simmons 1960; Turner and Rabalais 1994a) showed a reasonable agreement between the measured and reconstructed nutrient ratios. A similar reconstruction technique for the northern Adriatic Sea produced results that closely paralleled the real data (Justić et al. 1995a, 1995b). Comparison of measured and reconstructed nutrient ratios for the northern Gulf adjacent to the Mississippi River outflow revealed long-term changes in proportions of nutrients in the surface waters (Table 3.1). The reconstructed nutrient ratios for 1960 were farther removed from the Redfield ratio (Si:N:P = 16:16:1).

Probable nutrient limitation was assessed by Dortch and Whitledge (1992) and Justić et al. (1994), who compared the ambient nutrient concentrations with the k_s for nutrient uptake and, in the case of Si, a threshold value for uptake. Plots of relative frequencies (Figure 5.14) showed that dissolved N concentrations in the surface layer of the northern Gulf of Mexico during the period 1985–92 were lower than 1 μM in about 13% of the cases. Reactive P was below 0.1 μM in 17% of the cases, while reactive Si concentrations lower than 2 μM occurred in 25% of the cases. In contrast, the corresponding frequencies were 39%, 41%, and 10%, respectively, in 1960. These findings are important because studies of nutrient uptake kinetics (i.e., Rhee 1973; Har-rison et al. 1977; Goldman and Glibert 1983; Nelson and Brzezinski 1990) indicate that concentrations of 1 μM, 0.1 μM, and 2 μM may be considered as threshold values for N, P, and Si uptake, respectively. Thus, it appears that overall nitrogen and phosphorus nutrient limitation has decreased, while the probability of silicate limitation may have increased.

Fluctuations in the Si:N ratio within the riverine effluents and the offshore waters can affect diatom production and are believed to be major determinants in estuarine and coastal food web structure on a seasonal and annual basis, with major implications to oxygen and carbon cycling

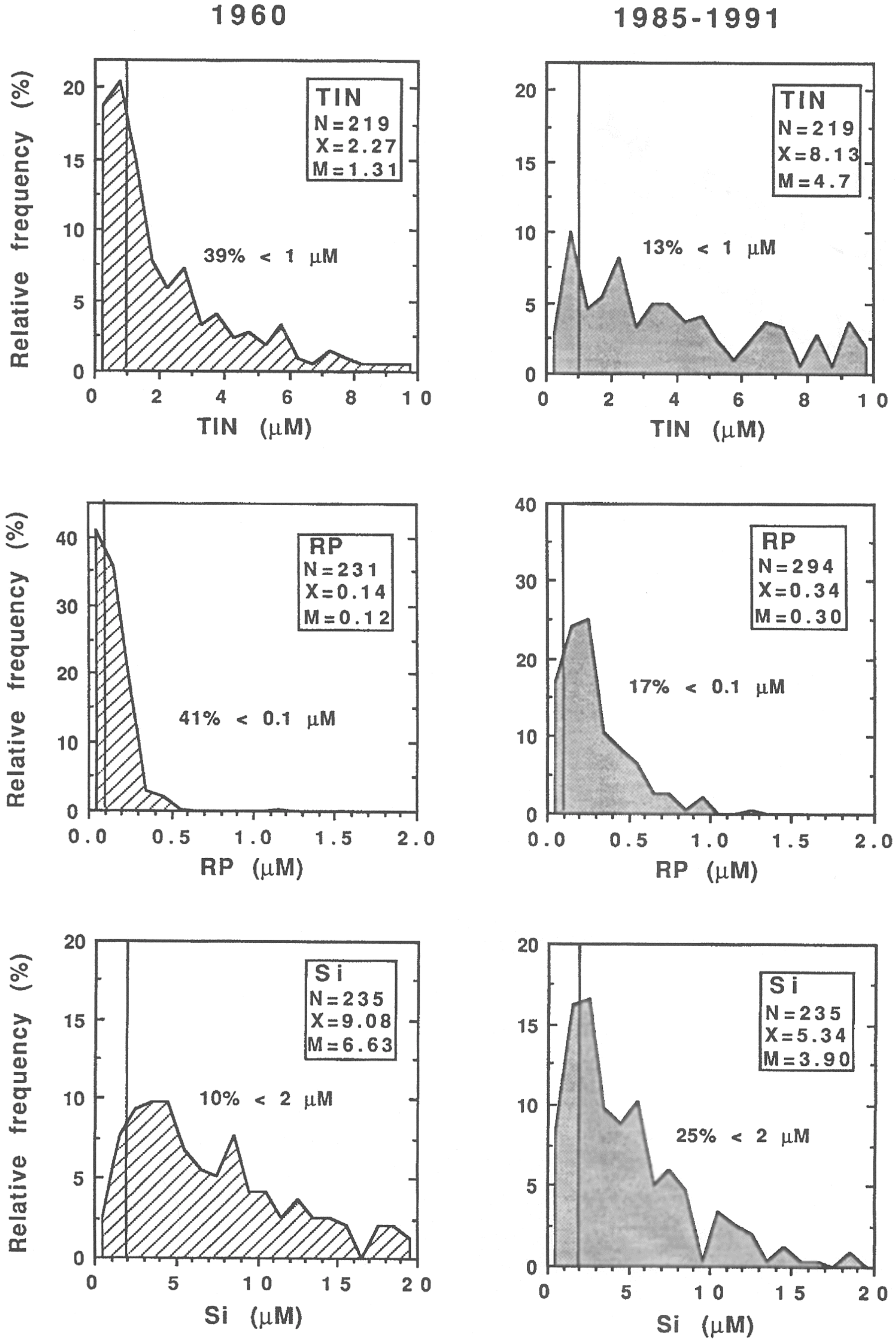


FIGURE 5.14. Relative-frequency polygons showing temporal changes in the surface nutrient structure of the northern Gulf of Mexico. NOTE: 1985–91 are field data; 1960 data are theoretical values calculated from the 1985–91 data (Justić et al. 1995b), assuming that the relative rates of change in nutrient concentrations over the intervening period were the same as in the Mississippi River (ΔN = +8.0%/yr, ΔP = +4.3%/yr, ΔSi = –.14%/yr). Vertical lines indicate threshold values for nutrient uptake based on literature data. (*From Justić et al. 1994.*)

(Turner et al. 1998). Deviations from the 16:16:1 Si:N:P ratio in nutrients available in the water column may be a limiting factor for diatoms, as well as for other phytoplankton groups (Hecky and Kilham 1988; Dortch and Whitledge 1992). Also, a decreasing Si:N ratio may exacerbate eutrophication by reducing the potential for diatom growth in favor of noxious flagellates (Officer and Ryther 1980). Another reasonable hypothesis that follows a more balanced nutrient composition, as evidenced in the Mississippi River and in the coastal waters as well, is that surface primary productivity has increased under these conditions.

5.5 ALLOCHTHONOUS ORGANIC C AND N INPUTS

Although the Mississippi River discharges organic matter, whose decomposition could consume oxygen in the coastal ecosystem, the principal source of organic matter reaching the bottom waters of the northern Gulf of Mexico influenced by the Mississippi River effluent and characterized by hypoxia is from *in situ* phytoplankton production (Rabalais et al. 1992b; Turner and Rabalais 1994b; Eadie et al. 1994; Justić et al. 1996, 1997). These results are verified by data from studies of the sedimentary environment, retrospective sediment analysis, empirical relationships of nutrient flux to primary production and net production, carbon flux studies, linkages of surface-water net production with oxygen uptake in the lower water column, and modeling experiments. The amount of organic loading in the Mississippi River is not large enough to account for the observed decline in oxygen over such a large area and volume (Turner and Allen 1982b). Also, the δ^{13}C signature of the particulate organic matter found in the sediments of the Mississippi River Bight indicates that terrestrial carbon sources are localized near the delta and nearshore sediments, and the bulk of carbon in the sediments of the hypoxic zone is from a marine origin (Rabalais et al. 1992b; Turner and Rabalais 1994b; Eadie et al. 1994; see Section 7.2).

Nutrient ratios of material flux from the Mississippi River also indicate that direct contributions of organic matter could account for much less of the sedimented carbon than marine phytoplankton production fueled by Mississippi River nutrients. Sedimenting marine phytoplankton generally have a C:N ratio (as atoms) of 9.5–9.9:1 (Meybeck 1982), whereas the C:N ratio of Mississippi River flux is about 2.3–3.7:1 (Leenheer 1982; Meybeck 1982; Telang et al. 1991; Trefry et al. 1994; Howarth et al. 1996). Therefore between 2.7 and 4.3 times as much organic matter (on average) could be supplied to marine hypoxic zones by marine algae grown with Mississippi River nitrogen than could be supplied directly by riverine organic matter. When the recycling of N within the coastal system is considered, the marine supply of organic matter would be greater than the allochthonous by an order of magnitude.

The decrease in suspended sediment load and associated organic nitrogen was discussed in Section 5.2. Mayer et al. (1998) pointed out that because the Mississippi River suspended sediment load has decreased (Meade 1995) and the dissolved inorganic nitrogen pool has increased (see Section 5.3), the relative importance of the organic nitrogen associated with the suspended sediment load might be less for the Mississippi River than for other large world rivers (Mayer et al. 1998). Thus, sediment-associated constituents may exacerbate hypoxia, but several lines of evidence show that sedimentation of marine organic matter resulting from increased river nutrients plays a preeminent role.

The contribution of nutrients and carbon from erosion of coastal wetlands in Louisiana has been proposed as a potential source of materials that fuels the development of hypoxia and its increase over time. Several lines of evidence discount this hypothesis. The land-loss rates were high from 1955/56 to 1978 (20% for marsh, equals a loss of 255,684 hectares; 16% for swamp, equals 32,730 hectares) and have decreased since then (Turner 1997). Indicators of eutrophication and worsening oxygen stress have continued to increase; thus there is no correlative link between land-loss rates and rates of eutrophication or increasing hypoxia. Carbon isotope signatures in nearshore sediments indicate that carbon emanating from marsh detritus is localized close to shore and does not become incorporated into the continental shelf food web (Fry et al. 1984; see also Section 7.2). Also, the amount of carbon released from erosion of wetlands is not sufficient to account for the observed decline in oxygen over such a large area and volume (similar to the allochthonous riverine carbon source argument, above), nor is the nitrogen released.

Estuarine exchanges with offshore waters clearly exist. Evidence for this is the inverse relationship between estuarine salinity and Mississippi River discharge (Wiseman et al. 1991). Therefore, the exchange of nutrients from inshore to offshore (and vice versa) is possible. If there were a significant dominance of nutrients in either direction, the sedimentary record of diatom production and eutrophication in estuarine and nearshore sediments would be similar (see Section 7.4). Biogenic silica (BSi) remnants in sediment cores from estuaries adjacent to the hypoxic zone (Turner et al. unpublished data) do not follow the same pattern of increases as offshore sediments (Turner and Rabalais 1994b). The accumulation rate of BSi in estuarine waters reflects the use of fertilizer in the estuarine basin, and the accumulation in offshore waters is coincidental with the nutrient loading of the Mississippi River. Thus, there is no coherence between nutrient loading in the estuarine and offshore waters, and offshore waters are dominated by *in situ* loading stimulated by Mississippi River nutrients and not estuarine nutrients.

5.6 OTHER CONSTITUENTS

The presence of multiple toxic chemicals in the effluent of the Mississippi River is perceived to be a potential cause for hypoxia in the Gulf. The carbon equivalents of these chemicals in the river's total dissolved and particulate carbon flux of insignificant and should not contribute to the depletion of oxygen. Also, some pollutants have increased this century, while others have declined in the last several decades (Meade 1995). Millie et al. (1994) suggested that the increased atrazine (herbicide) levels in the 1993 flood discharge (Goolsby 1994) could be detrimental to phytoplankton growth, as demonstrated in laboratory experiments. Although atrazine flux increased during the 1993 flood, the abundance and biomass of cyanobacteria (phytoplankton affected in laboratory experiments) increased (Dortch 1994). Therefore, concerns about the increased flux of atrazine were not supported by the 1993 picocyanobacterial abundance.

Time-series analysis disclosed significant declines in pH and alkalinity and increases in strong acid anions (phosphoric, nitric and sulfuric acid) in the lower 300 km of the Mississippi River (Bryan et al. 1992). In surface waters of the Gulf inner shelf, pH levels vary mostly between 8.1 and 8.5, with no clear seasonal signal or gradient with distance from the river effluents. The large, natural buffering capacity of seawater makes the effects of pH changes in the Mississippi River unlikely to be an effect in the open Gulf.

CHAPTER 6

Biological Processes

Because there is great daily, weekly, and seasonal variability in current flow and stratification on the shelf, there is no simple description of the couplings between nutrient delivery, carbon production in surface waters, and delivery and recycling in bottom waters. There is, however, evidence of an ecological "signal" (couplings) amid the "noise" (the variability), when more extensive data sets and longer time periods are examined. These couplings implicate changes in riverine nutrients and nutrient ratios with the overall effects on productivity, carbon accumulation at the seabed, and low oxygen conditions on this shelf.

The data synthesized in this chapter describe how the coastal ecosystem functions and the components within that ecosystem relevant to hypoxia are drawn from detailed hydrographic, chemical, and biological data; empirical relationships; process studies; and modeling experiments. The data provide information on a variety of scales, from experiments for a parcel of water from a particular locale over a limited time, to more integrative measures of ecosystem response and change over broader spatial and temporal scales. The data are drawn from the area of concern on the Louisiana continental shelf and from similar ecosystems worldwide.

6.1 DILUTION, UPTAKE, AND REGENERATION OF NUTRIENTS

Despite the extremely high nutrient inputs to the shelf, nutrients are depleted to low, and sometimes undetectable, concentrations within a short distance of the river mouth (Lohrenz et al. 1990, 1997, 1999; Rabalais et al. 1991, 1996, 1998; Dortch and Whitledge 1992; Nelson and Dortch 1996). Nitrate, the major form of nitrogen supplied by the river, decreases quickly. Consequently ammonium, resupplied by regeneration (Bode and Dortch 1996), is the only available nitrogen source over much of the shelf, but it is present at much lower concentrations than nitrate is initially. Dissolved organic nitrogen (DON) is supplied by the river, but its dynamics in the outflow of the Mississippi River are complex, and it is unclear if it is a source or a sink for nitrogen available to phytoplankton (Lopez–Veneroni and Cifuentes 1994). Silicate can be depleted to extremely low levels. In fact some of the lowest concentrations of silicate observed in any ocean were measured on the Louisiana shelf (Nelson and Dortch 1996). Dissolved inorganic phosphate concentrations also decrease to low levels, but are usually detectable. The importance of dissolved organic phosphorus has not been fully assessed, although it may be recycled rapidly (Ammerman 1992; Ammerman et al. 1995). The area with high nutrient concentrations varies, depending on river flow and season, but is much smaller than the total area of the Louisiana–Texas shelf affected by the Mississippi River. High phytoplankton biomass and productivity can be maintained due to high rates of nitrogen and phosphorus regeneration (Ammerman 1992; Gardner et al. 1994; Ammerman et al. 1995; Bode and Dortch 1996). Short-term rates of Si regeneration are generally much lower, which increases the potential for Si limitation.

The rapid depletion of nutrients is due to biological uptake and conservative mixing between high-nutrient Mississippi river water and low-nutrient, high-salinity Gulf of Mexico water, with the relative importance of these processes depending on the season and river flow. At most times there are large negative deviations from the conservative mixing line in estuarine mixing diagrams of nutrient concentration vs. salinity, which are indicative of high biological removal (Figure 4.2; Dortch and Whitledge 1992; Turner and Rabalais 1994a; Nelson and Dortch 1996; Lohrenz et al. 1999). Rapid rates of nitrogen and silicate uptake have been confirmed by direct measurements (Bode and Dortch 1996; Nelson and Dortch 1996).

Nonlinear mixing diagrams (Lohrenz et al. 1999) suggest other sources of nutrients, such as low-salinity nutrient sources (desorption from sediments or production of dissolved organic nitrogen) or intrusion of deeper offshore nutrients (see Section 8.1.1) (Fox et al. 1985; Lopez–Veneroni and Cifuentes 1994). Although the N:P ratio in river water is near the Redfield ratio and suggests that either may be limiting primary production in receiving Gulf waters, the efficient remineralization of P in marine systems coupled with the inevitable losses of N due to denitrification probably means that N is the limiting nutrient (of N and P) beyond the immediate plume and may explain some of the nonlinear mixing diagrams in Fox et al. (1985), Hitchcock et al. (1997), and Lohrenz et al. (1999).

6.2 NUTRIENT-ENHANCED PRIMARY PRODUCTION

High biological productivity in the immediate (320 g C m^2/yr) and extended plume (290 g C m^2/yr) of the Mississippi River (Lohrenz et al. 1990; Sklar and Turner 1981; respectively) is mediated by high nutrient inputs and regeneration and by favorable light conditions. Small-scale and short-term variability in productivity are the consequence of various factors, such as nutrient concentrations, temperature, salinity, and light (Lohrenz et al. 1990, 1994), but are also clearly influenced by Mississippi River flow and nutrient flux to the system (Justić et al. 1993, 1997; Redalje et al. 1994; Lohrenz et al. 1997; Rabalais and Turner 1998). "New" nutrients become depleted along the river-to-ocean mixing gradient through dilution and biological uptake, and regenerated water column nutrients support primary production for great distances from the river mouth (Bode and Dortch 1996; Nelson and Dortch 1996). Additional nutrients become available through regenerative processes in the sediments (see Section 6.10). Surplus nitrogen is considered the primary cause of coastal eutrophication in most systems, including the Mississippi River, but there is evidence for limitation by phosphorus and silica at times (Ammerman 1992; Nelson and Dortch 1996; see Section 6.3). In addition to the role of nutrients, the importance of other factors, such as light availability and mixing rates, must be considered in attempts to understand the temporal and spatial patterns in primary production.

Lohrenz et al. (1997) clearly demonstrated that primary production in shelf waters near the delta and some distance from it was significantly correlated with nitrate and nitrite concentrations and fluxes over the six-year period 1988–94 (Figure 6.1). Light limitation was likely an important factor during winter months, but a positive correlation was demonstrated between river inputs of nitrate and nitrite from other times of the year. The relationships between riverine flux and concentration for those stations on the western end of their study area (i.e., near the transect C study area of Rabalais, Turner, and Wiseman, as shown in Figure 2.1) were improved when the riverine input data were lagged one month. These results were consistent with those of Justić et al. (1993, 1997) for a one-month lag between net production in surface waters and river discharge and nitrate flux. Even stronger correlations were observed between the concentration of orthophosphate and primary production, but these were not significant (smaller sample size).

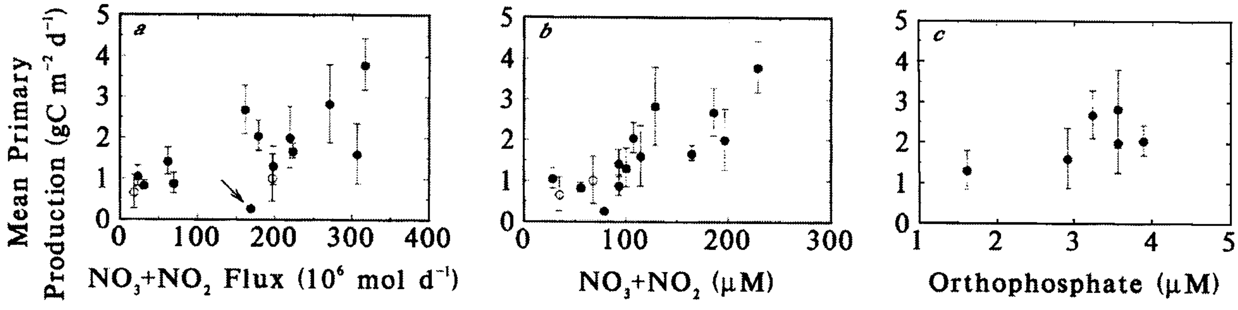


FIGURE 6.1. Relationship between mean primary production for the combined central and eastern regions of the Mississippi River Bight and (a) river-borne nitrate and nitrite flux, (b) nitrate and nitrite concentration at Venice, and (c) orthophosphate concentrations at Belle Chase. NOTE: Historical primary production data from Thomas and Simmons (1960) are included in (a) and (b) for comparison (o). Error bars are ± 1 s.e. (*From Lohrenz et al. 1997.*)

There was a high degree of coherence between Mississippi River nitrate fluxes and net production rates at Station C6* for 1985–92 (Justić et al. 1997). Mississippi River discharge for 1985–92 had a typical sinusoidal seasonal pattern consistent with longer records (Bratkovich et al. 1994). Riverine nitrate flux was highest in April and lowest in September, also consistent with Bratkovich et al. (1994). Although the monthly cycle of Mississippi River nitrate flux generally resembled the monthly cycle of freshwater runoff, the peak in nitrate flux was somewhat delayed with respect to the peak in freshwater runoff and was the result of a seasonal pattern in riverine nitrate concentrations (Figure 6.2). The net production rates for the upper water column (Justić et al. 1997) showed a well-defined seasonal cycle, with a minimum of –0.2 g C m^2/d in December and a maximum of 1.2 g C/m^2/d in April. A cross-correlation function of net production and riverine nitrate flux showed that the two series were highly correlated (CCF = 0.73, $p < 0.01$) and that a seasonal maximum in net production lags with respect to a riverine nitrate maximum by one month (Figure 6.3).

Maximum values of biomass (Rabalais and Turner 1998) and primary production (Lohrenz et al. 1990, 1999) are typically observed at intermediate salinities and coincide with nonconservative decreases in nutrients along the salinity gradient (i.e., biological uptake). Surface chlorophyll *a* concentrations peak at intermediate salinities where light conditions improve due to decreased suspended sediment concentrations and where nutrient concentrations are sufficient for phytoplankton growth (Figure 6.4). There is a consistent westward and downstream transition away from the discharges of the Mississippi and Atchafalaya Rivers along the coastal plume in lower to higher salinities, higher to lower nutrients, and higher to lower surface chlorophyll concentrations (Rabalais and Turner 1998) (Figures 6.5–6.7 cf. Figure 4.3). Ultimately, nutrients become limiting to phytoplankton productivity. These gradients away from the riverine sources are further reflected in the flux of organic material as seen in surface-to-bottom pigment ratios and accumulation of phaeopigments in the lower water column. Respiration rates are higher in shallower waters and are related to chlorophyll *a* concentrations (Turner and Allen 1982b; Turner et al. 1998; Turner and Rabalais 1998a). Therefore, there is a consistent transition away from the river discharges along the coastal plume in flux of organic material, respiration rates, and incidence of bottom-water hypoxia.

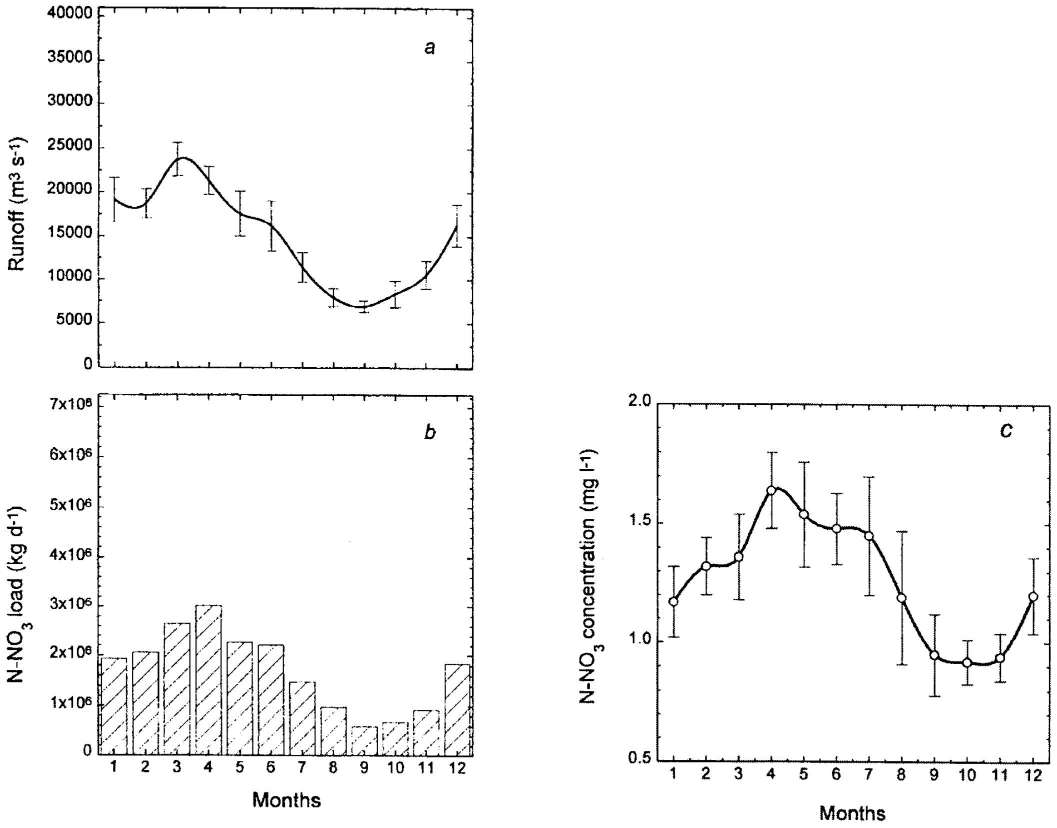


FIGURE 6.2. Monthly averages (1985–92) of Mississippi River runoff at Tarbert Landing (a), Mississippi nitrate flux at Tarbert Landing (b), and Mississippi nitrate concentration at St. Francisville (c). NOTE: Vertical bars represent ± 1 s.e. *(Modified from Justić et al. 1997.)*

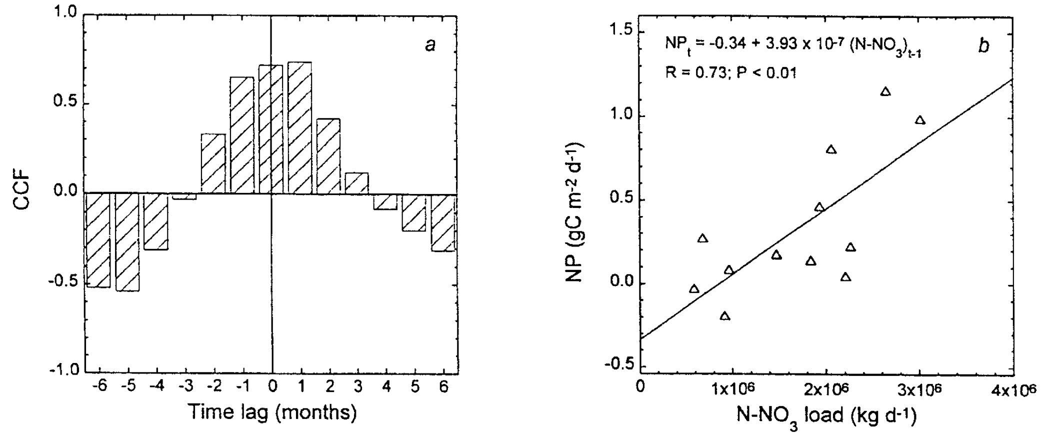


FIGURE 6.3. Left panel: Cross-correlation function (CCF) for Mississippi River nitrate flux at Tarbert Landing and net productivity of the upper water column (1–10 m at station C6*). Right panel: Best-fit time-delayed linear model for the regression of net production (NP) on nitrate load. NOTE: The model is $NP_t = 0.34 + 3.93 \times 10^{-7}$ nitrate $_{t-1}$ where t and t-1 denote values for the current and preceding months, respectively. Symbols denote monthly averages for the period 1985–92. *(From Justić et al. 1997.)*

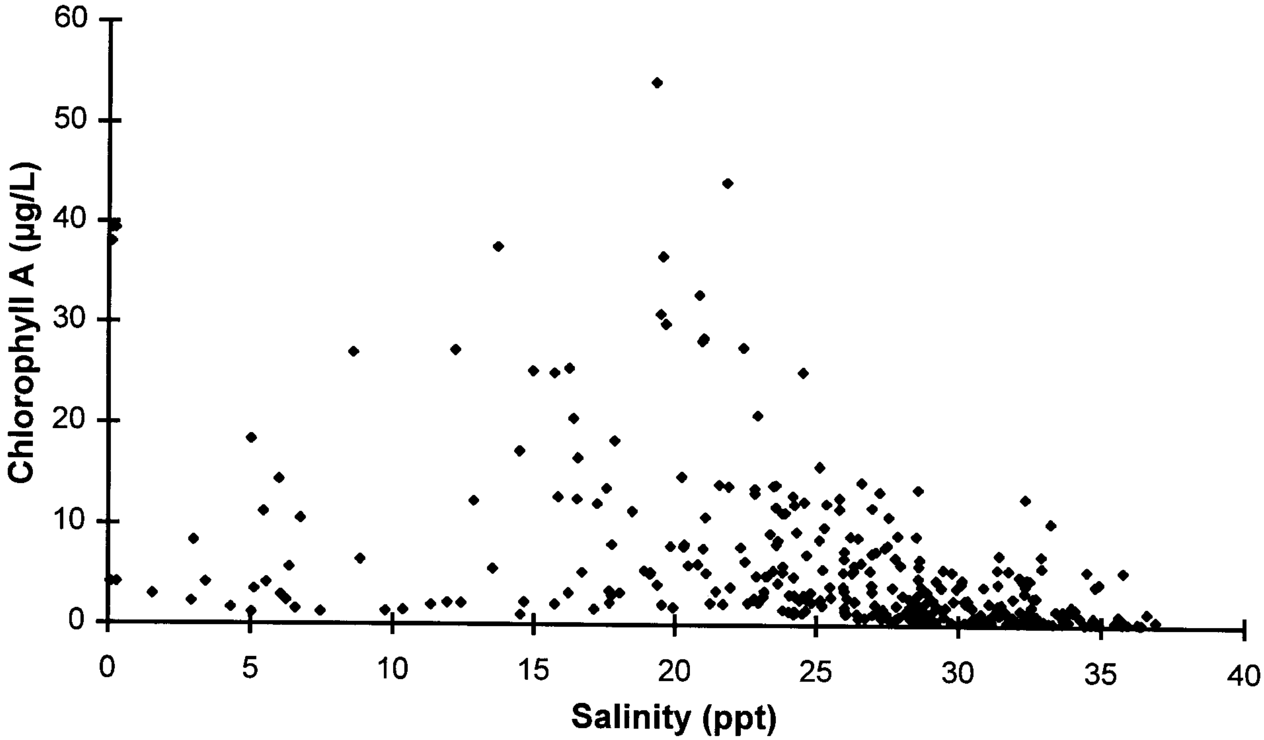


FIGURE 6.4. Relationship of surface chlorophyll *a* and salinity for six LATEX cruises (Apr. '92, Oct. '92, Apr. '93, Jul. '93, Apr. '94, Jul. '94) from 89.5°W to 97°W. NOTE: Maximum value of 209 µg/l was deleted from the plot. (*From Rabalais and Turner 1998.)*

The distribution of low dissolved oxygen in bottom waters is related to high surface net production, but this relationship is lagged in time and space (Justić et al. 1993; Rabalais et al. 1994). Phytoplankton and fecal pellets in surface waters fall into the bottom layers quickly, perhaps in a few days or less (see Sections 6.5 and 6.6), but surface and bottom currents are not traveling in the same direction or at the same speed. Most of the organic matter reaching the bottom is consumed in many days (and probably weeks), rather than hours (see Section 6.7 on respiration). The depletion of oxygen is cumulative and depends also on the reaeration rate. Therefore, no consistent patterns should be expected between surface phytoplankton pigments in surface waters and low oxygen in bottom waters for one specific location in time. This is contrary to the pattern reported in Leming and Stuntz (1984), who suggested that satellite imagery of surface-water chlorophyll levels could describe the distribution of hypoxia. The lack of such a pattern was verified by Rabalais et al. (1991) and Rabalais (1998b) with several data sets (Figure 6.8).

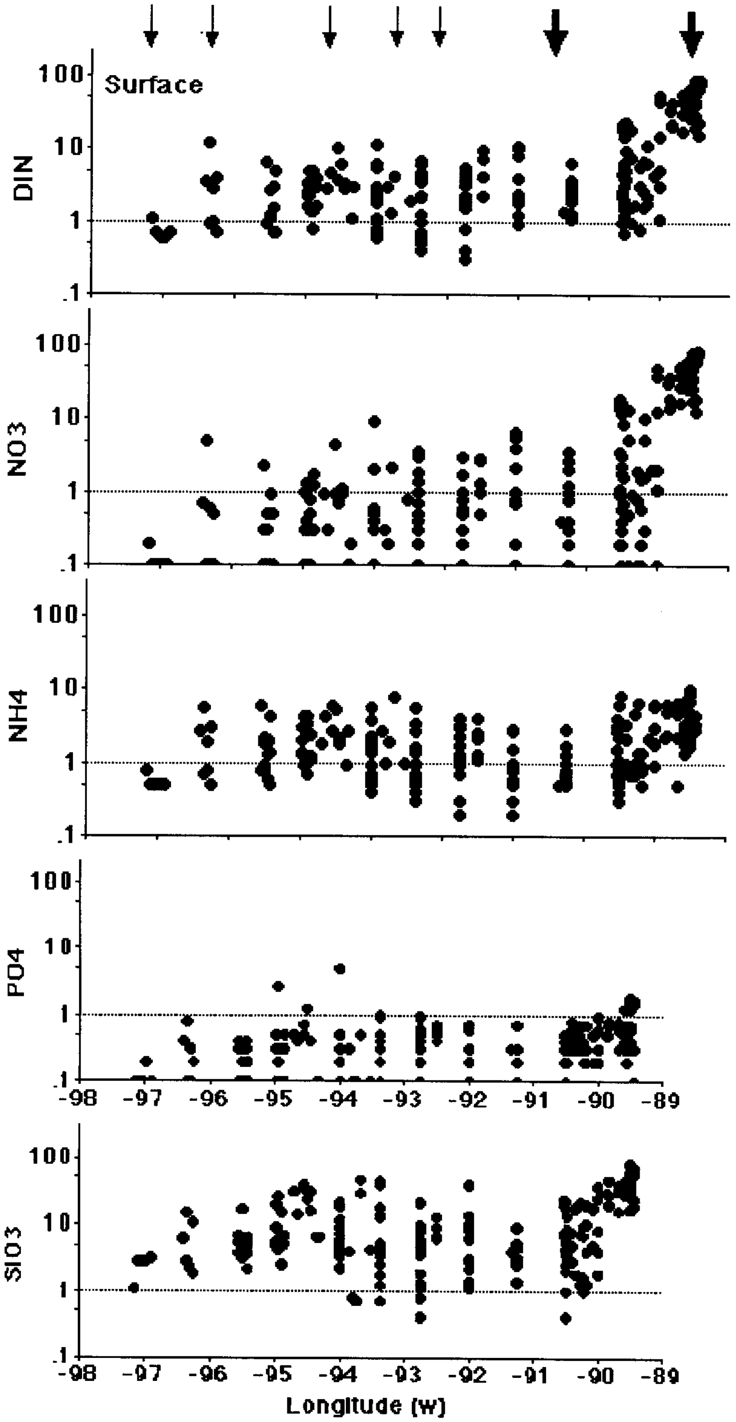


FIGURE 6.5. Concentration of dissolved inorganic nutrients (μM) in surface waters at the LATEX sampling stations (Figures 3.4 and 3.5). NOTE: Total depth at these stations was 10–100 m. The arrows indicate the source and relative contributions of freshwater inputs. (*From Rabalais and Turner 1998.*)

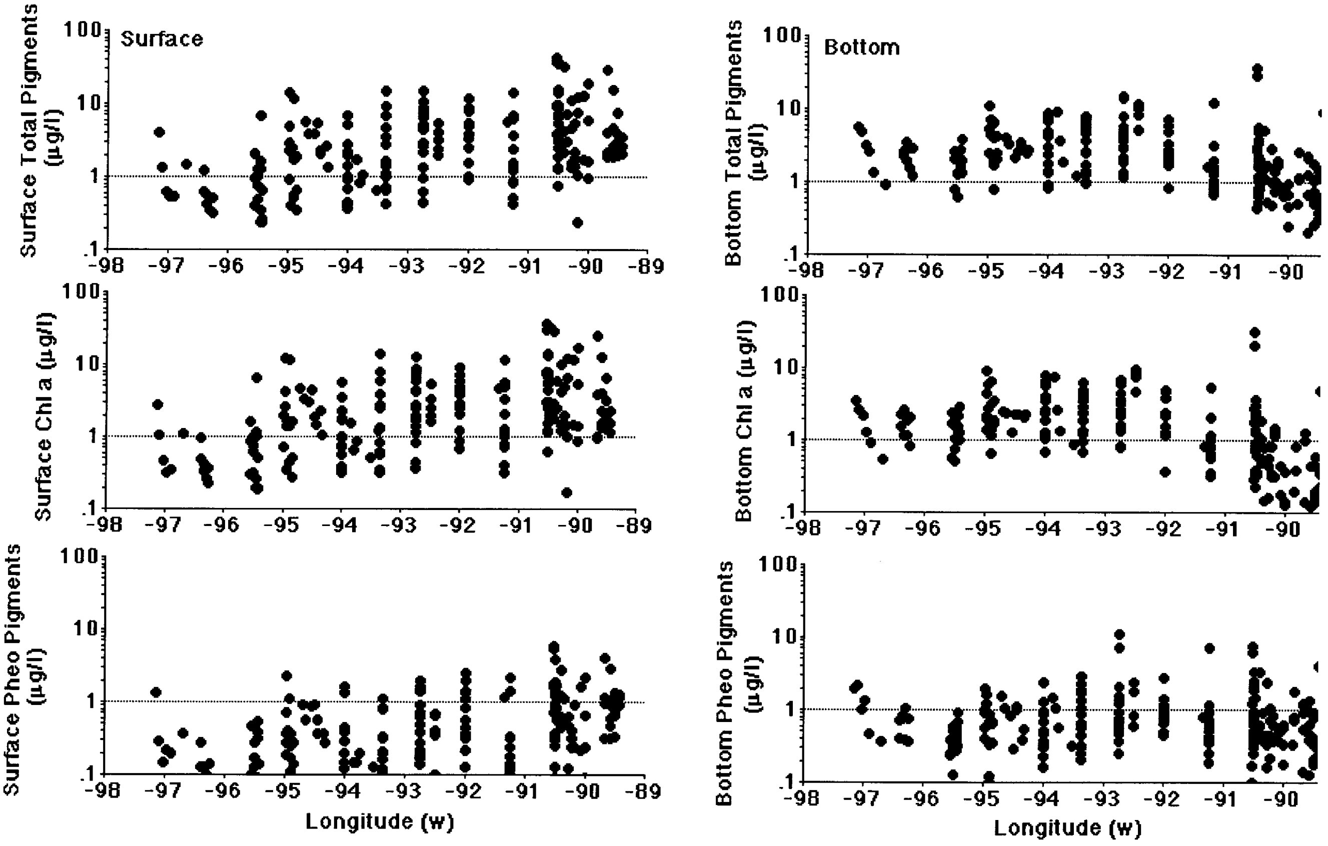


FIGURE 6.6. Relationship between the concentration of phytoplankton pigments (total, chlorophyll *a*, and phaeopigments (µg/l)) and longitude in surface and bottom waters at the LATEX sampling stations (Figures 3.4 and 3.5). NOTE: Total depth at these stations was 10–100 m. (*From Rabalais and Turner 1998.*)

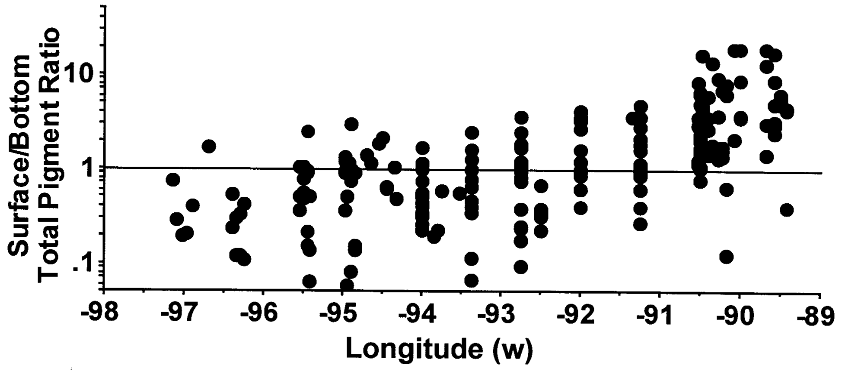


FIGURE 6.7. Relationship between the ratio of the concentration of total phytoplankton pigments (µg/l) in surface to bottom waters with longitude at the LATEX sampling stations (Figures 3.4 and 3.5). NOTE: Total depth at these stations was 10–100 m. (*From Rabalais and Turner 1998.*)

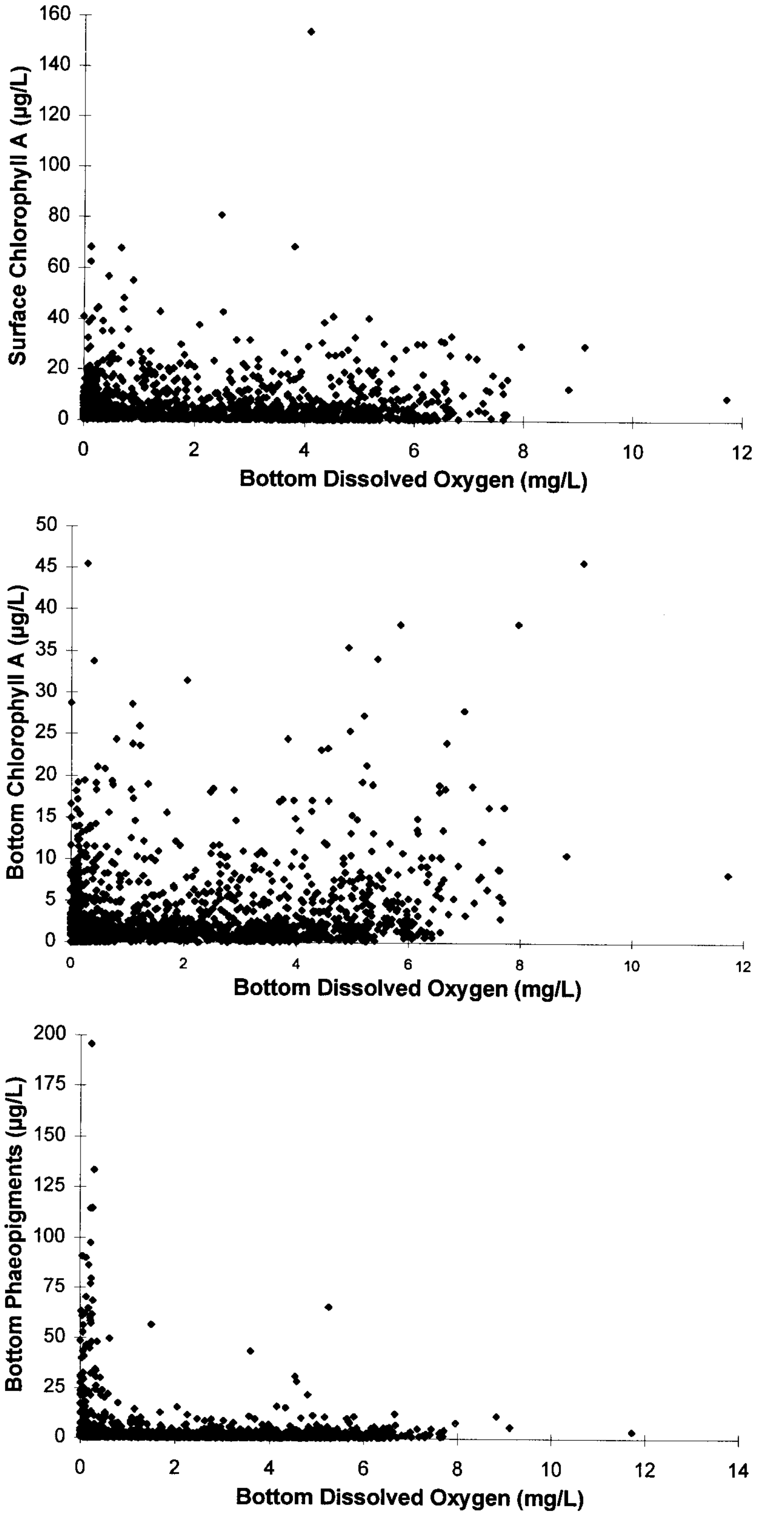


FIGURE 6.8. Comparison of bottom-water dissolved oxygen with surface-water chlorophyll *a*, bottom-water chlorophyll *a*, and bottom-water phaeopigments for all stations and all cruises May–September 1985–96. (*Rabalais unpublished data.*)

There is also no clear pattern between hypoxia and the concentration of chlorophyll *a* in bottom waters. One might predict that viable phytoplankton in bottom waters would produce oxygen at a sufficient rate to offset consumptive processes and thus prevent anoxia (Dortch et al. 1994). A compilation of data (Figure 6.8) reveals that hypoxic bottom waters may have negligible chlorophyll *a* levels, but others ranged up to 45 µg/l. In many instances, elevated phaeopigments in hypoxic waters were usually associated with very low oxygen concentrations (< 0.5 mg/l).

6.3 NUTRIENT LIMITATION OF PHYTOPLANKTON GROWTH

Patterns of nutrient depletion provide evidence that riverine inputs of nutrients and their pattern of regeneration ultimately limit the extent of river-enhanced productivity and biomass. The nutrient most relevant to the overall production of phytoplankton production over the broad region fueling hypoxia is nitrogen. It follows, and is supported with evidence from long-term data sets (Turner and Rabalais 1994a) and the sedimentary record (Turner and Rabalais 1994b; Eadie et al. 1994), that increases in riverine dissolved inorganic nitrogen loads are highly correlated with indicators of increased productivity in the overlying water column, i.e. eutrophication of the continental shelf waters. Phosphorus and silicon limitation have been identified as well, along with multiple limitations of combinations of N, P, and Si. The importance of understanding nutrient limitation is that nutrient management in the watershed, constituents, and timing need to be related to biological processes on the shelf and long-term ecosystem responses.

A variety of methods (not without their inherent problems) have been used to determine which nutrients are limiting and to show the temporal and spatial variations in their limitation (Table 6.1). The most widely used indicator is the simultaneous comparison of ambient nutrient concentrations and ratios (Table 6.1; Justić et al. 1994, 1995b; Turner 1998b). A nutrient is considered limiting if its concentration is less than a generally accepted affinity coefficient (k_s) for its uptake and if ratios of concentrations between all nutrients indicate that nutrient is in shortest supply by comparison with the Redfield ratios (Fisher et al. 1988; Dortch and Whitledge 1992). Additional information has been provided by bioassay experiments in which possible limiting nutrients are added singly and in combination with natural plankton (Smith and Hitchcock 1994; Dortch et al. unpublished data; Turner unpublished data) or phytoplankton cultures in filtered natural sea water (Smith and Hitchcock 1994). Growth over time is monitored by fluorescence or cell counts. The limiting nutrient or nutrients are those that stimulate the growth of all phytoplankton or particular groups of phytoplankton in comparison with controls.

Although no single approach can give a definitive answer, some understanding of this system's nutrient limitation has emerged from the following multiplicity of approaches (Table 6.1):

1. P and Si limitation, and sometimes multiple limitation, have all been identified at many times and places in this region (Table 6.1; Justić et al. 1994, 1995b; Turner unpublished data). This is consistent with the hypothesis of Justić et al. (1995b) that the system is in stoichiometric balance, so that any nutrient can become limiting. What Table 6.1 does not show is that the area where nutrient limitation is observed in any period is highly variable and can range from most stations sampled to only a few

2. Most indicators of nutrient limitation give results consistent with other indicators, but there are some discrepancies. For example, in July 1993 nutrient concentrations and ratios indicated overwhelming P limitation, but alkaline phosphatase assays did not.

TABLE 6.1. Summary of studies identifying nutrient limitation along the Louisiana coast influenced by the Mississippi River.

Method	Cruise Dates									
	July/Aug. 1987	***April 1988***	***Sept. 1989***	***July/Aug. 1990***	***March 1991***	***Sept. 1992***	***May 1992***	***July 1992***	***Apr./May 1993***	***July 1993***
Nutrient Concentrations and Ratios	Si=N>P[4]	Si>P>N[4, 8]	N>Si [8]	Si at Low S & Si or N at High S[3]; P>>N=Si[8]	P>>Si>N[8]; P & Si[10]	N>>P=Si[8] N & Si[10]	P >Si[8]	N[5]	Si[5]	P>>Si= N[6]
Fluorescence Bioassays, Natural Plankton					P & Si[10]	N & Si at High S[10]	P & Si[10]			Not Si or N[6]
Cell Count Bioassays, Natural Plankton										Diatoms: Si & N[6]
Fluorescence Bioassays, Cultured Phytoplankton						N & Si at High S[10]	P & Si[10]			
Phytoplankton Species Composition					Diatoms Si Limited at High S[3, 7]					
Taxon-Specific Growth Rates					Growth all phytoplankton correlated with N & Si[7]					
Alkaline Phosphatase					P at Low S[1]		Not P[1]	Not P[2]		Not P[2]
Phosphate Turnover					P at Low S[1]		Not P[1]			
Amino Acid/Protein Ratio	Some N	Some N								
Si Uptake Kinetics								Not Si[9]	Si at High S[9]	

NOTE: Relative ranking of frequency of limitation shown where possible by > and =. Some studies focus on limitation by only one or two nutrients, indicated in parentheses after sources: P = phosphorus, N = nitrogen, Si = silicate, and S = salinity.

Sources: [1]Ammerman 1992 (P); [2]Ammerman et al. 1995 (P); [3]Dortch et al. 1992b (Si & N); [4]Dortch & Whitledge 1992; [5]Dortch et al. 1995 (Si & N), [6]Dortch et al. unpublished data (Si & N); [7]Fahnenstiel et al. 1995 (Si & N); [8]Lohrenz et al. 1999; [9]Nelson and Dortch 1996 (Si); [10]Smith & Hitchcock 1994.

3. N generally limits overall productivity in this system (Turner and Rabalais 1994b; Lohrenz et al. 1999). N limitation occurs most often at higher salinities and during low-flow periods.
4. P limitation is much more likely than was originally expected. It occurs most often at intermediate salinities and during periods of high freshwater input, as in other estuarine systems (Fisher et al. 1995). Although high flow and P limitation would usually occur in the spring in the plume front, extraordinarily high flow in the late spring of 1993 resulted in some indicators suggesting pervasive P limitation over a large area during that summer.
5. Si limitation obviously affects diatoms, the only abundant siliceous autotrophs in this system, but diatoms are a large fraction of the biomass (Dortch 1998). The occurrence of Si limitation appears to be more spatially and temporally variable than P or N limitation. Dortch and Whitledge (1992) initially observed, and Nelson and Dortch (1996) confirmed, that Si limitation was more prevalent in spring than summer. However, the data in Table 6.1 indicate that Si limitation of diatoms does occur in the summer, even in years when river flow is not anomalously high (for example, 1994). Further, severe Si depletion was observed in the late fall and winter of 1997, perhaps after cold-front passage followed by sunny, calm weather that stimulated diatom blooms (Parsons and Dortch unpublished data). Clearly, the prevalence of Si limitation is not easily predicted and requires methods targeted specifically at diatoms.
6. Not all phytoplankton are limited by the same nutrient. This is intuitively obvious for differences between diatoms and nondiatoms, but also occurs at the species level.
7. There is considerable interannual variation in the degree, kind, and seasonality of nutrient limitation, which is related to variations in riverine input, but also to conditions and weather in the outflow area.

Besides the usual criticisms of using nutrient concentrations and ratios to assess limitation, a methodological caveat is necessary for this region. Almost all of the nutrient analyses were conducted on unfiltered samples. There is a debate over the need for filtration. Some believe that filtration results in contamination or loss due to adsorption onto filter holders. Others believe that nutrients bound to sediments or biogenic silica are released by chemicals used in the assays, resulting in higher concentrations of nutrients, which may or may not be biologically available.

Because of the impact that sediment-adsorbed nutrients would have on assessing nutrient limitation in turbid estuaries, Dortch et al. (unpublished data) examined this problem for ammonium and silicate. They found that, on average, ammonium concentrations in filtered samples were 60% of those in unfiltered samples, but that the variation around 60% was so great that a single correction factor could not be applied. The cause of the discrepancy was not contamination, adsorptive losses during analysis, or ammonium bound to particles, but solely light absorption by particles in samples.

Although this is correctable in future analyses, it means that ammonium concentrations in older data sets may be overestimated. Because residual ammonium keeps dissolved inorganic nitrogen (DIN) concentrations and DIN/P and Si/DIN ratios from being low enough to indicate N limitation, overestimating ammonium could have underestimated N limitation. Phosphate analyses may also be subject to the same problem (Fisher et al. 1995). For silicate analyses, the differences between filtered and unfiltered samples were much less (Dortch et al. unpublished data). Most silicate analyses for waters of the hypoxic area have been conducted with assays optimized for the large range of concentrations encountered and have not been optimal for measuring the low concentrations that signal Si limitation (Nelson and Dortch 1996). Consequently, Si limitation may also have been underestimated.

Riverine nutrient inputs, ambient nutrient concentrations and ratios, bioassay experiments, and other indicators of nutrient limitation all suggest that N, P, or Si may be limiting at some times and places in the outflow of the Mississippi River. The details of when and where particular nutrients are limiting and the severity of limitation are not completely known. P limitation, however, occurs mostly at a lower salinity and is probably a factor mainly in the mixing zones of the river plumes. N is the nutrient limiting production overall and the nutrient for which loading has increased in recent decades. Lohrenz et al. (1999) suggested that although N is the overall limiting nutrient, reduction of diatom blooms in the plume front can be accomplished by decreasing phosphorus loading. This area, however, is not the only—or necessarily the prime—area for source of organic deposition to the benthic layers over the broader hypoxic area.

6.4 PHYTOPLANKTON SPECIES COMPOSITION AND CHANGES IN RESPONSE TO NUTRIENT LIMITATION

Phytoplankton in the region can be divided into three groups: picocyanobacteria, diatoms, and others (Table 6.2, Figure 6.9). Others include mostly small flagellates, although dinoflagellate blooms do occur sporadically (Dortch 1998; Dortch et al. accepted). Although the small flagellates can be numerically abundant, they are much less abundant than picocyanobacteria and so much smaller than most diatoms that they are unlikely to dominate the biomass. Further, there is relatively little seasonal variation in flagellate abundance (Figure 6.9). Thus, picocyanobacteria and diatoms are the most important phytoplankton groups in the hypoxic region. Picocyanobacteria are most abundant in the summer and early fall, whereas diatoms are most abundant in the spring (Table 6.2, Figure 6.9). In terms of estimated biomass, diatom biomass usually dominates, especially in the spring. The exception was during the flood in July 1993, when all phytoplankton were elevated, but especially picocyanobacteria (Rabalais et al. 1998; see Section 3.7).

TABLE 6.2. Average abundance, % relative abundance (Abundance Group/Abundance Total Phytoplankton x 100), and estimated biomass for six cruises along the Louisiana–Texas shelf.

	April 1992	October 1992	April 1993	July 1993	April 1994	July 1994
Total Phytoplankton						
Abundance (#Cells/1 x 10^6)	84.60	226.00	34.00	721.00	59.30	318.00
% Phytoplankton						
Diatoms	6.97	0.38	9.75	0.12	10.26	1.00
Picocyanobacteria	82.90	95.33	74.52	97.12	72.87	95.94
Other	10.13	4.29	15.73	2.76	16.87	3.06
Estimated Biomass (µgC/liter)						
Diatoms	200.00	33.00	127.00	14.90	600.00	36.00
Picocyanobacteria	7.11	21.00	2.80	69.00	4.74	30.30

Note: The number of samples ranged from 79 in April and October 1992 to 124 in July 1994.
Source: From Dortch 1998.

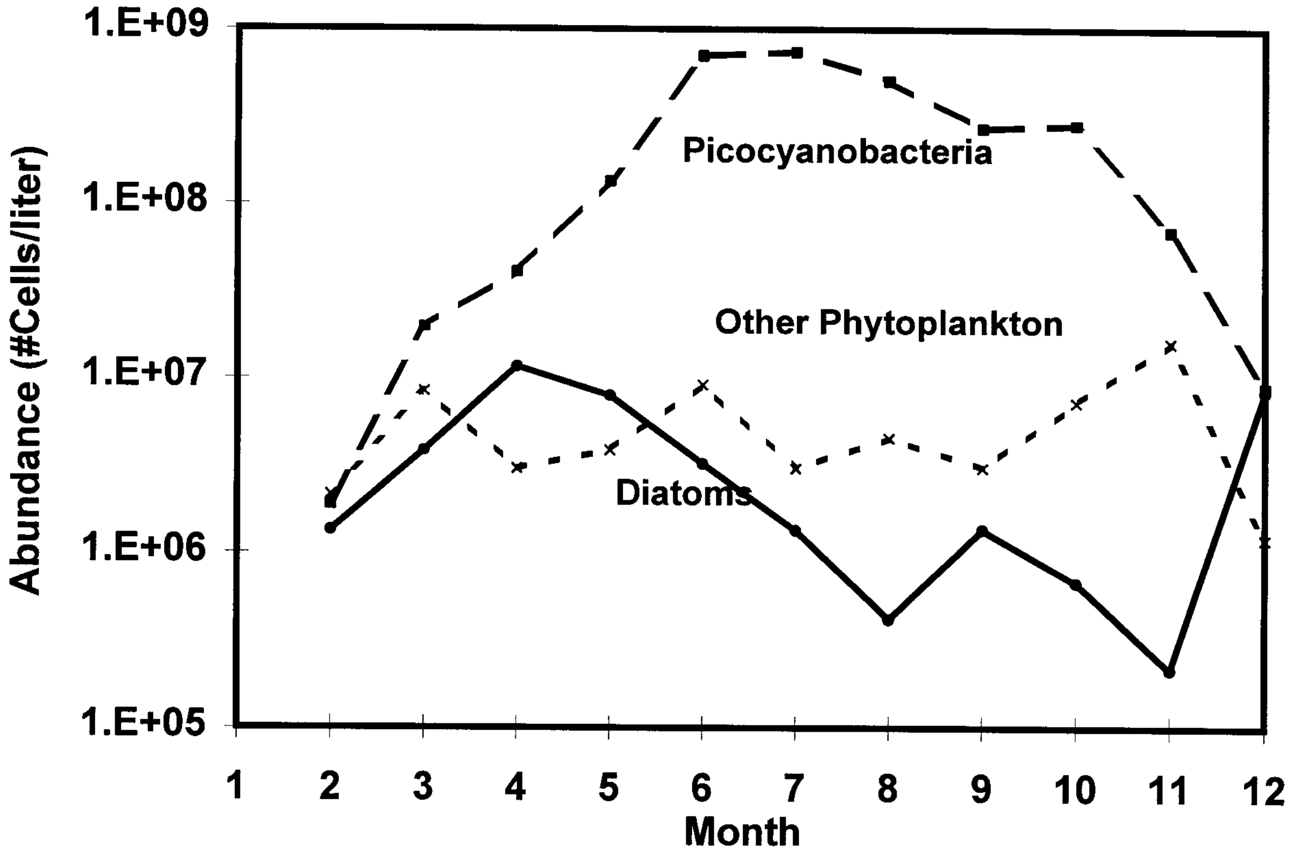


FIGURE 6.9. Average monthly surface layer phytoplankton major taxa composition at C6A and C6B from 1990 to 1995. NOTE: Number of samples ranges from 14 to 41 for all months, except February (n = 5) and December (n = 2). (*Dortch unpublished data.*)

On the Louisiana shelf in the area influenced by the Mississippi River, the prevalence of Si limitation, indicated by nutrient concentrations and ratios (Table 6.1), suggests that diatoms may at times be Si limited, whereas nondiatoms are not. Several lines of evidence support this contention:

- The annual average abundance of diatoms is inversely proportional to the prevalence of Si limitation in the same year (Figure 6.10), except 1992, which was unusual for other reasons.
- Cell count bioassay experiments show that diatoms are limited by Si, N, or Si + N availability, sometimes when other phytoplankton are limited by another factor.
- A shortage of silicate does not just modify the relative abundance of diatoms and nondiatoms. It also influences the diatom species composition so that when Si is limiting, diatoms with low Si requirements, as manifested by light silicification of the frustule (Dortch et al. 1992b; Fahnenstiel et al. 1995) or lower k_s for silicate uptake (Nelson and Dortch 1996), predominate.

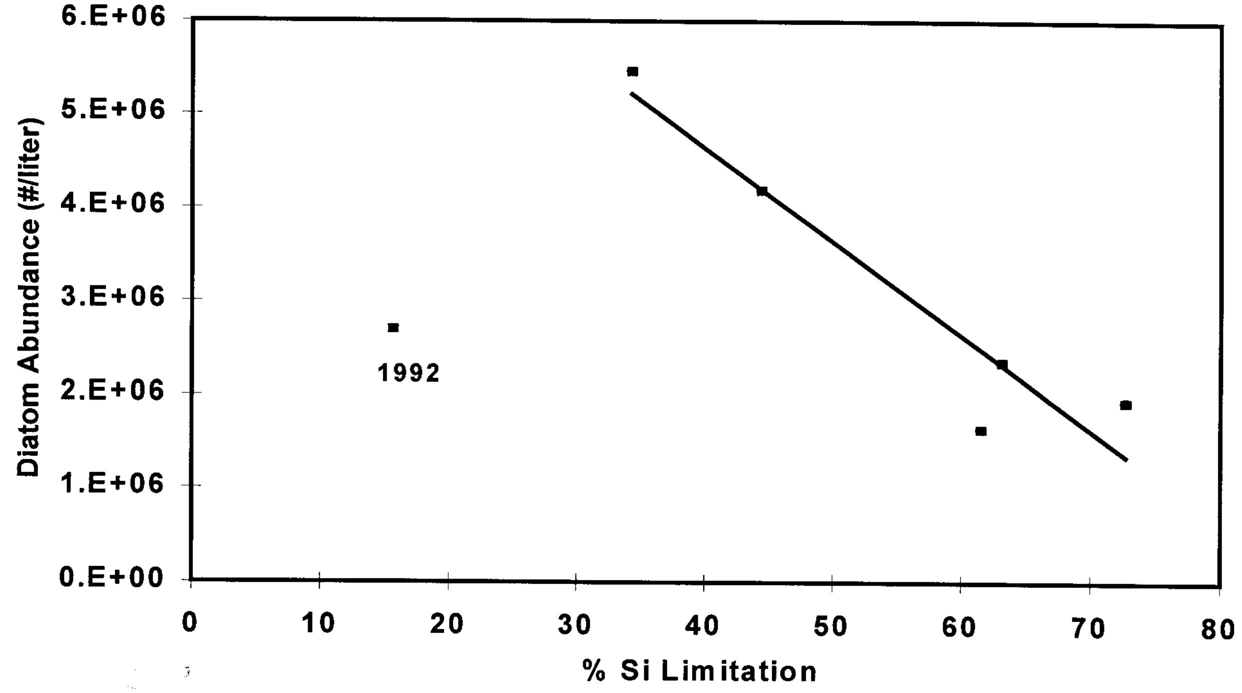


FIGURE 6.10. Average annual diatom abundance versus average annual % Si limitation in the surface layer at C6A and C6B from 1990 to 1995. NOTE: The % Si Limitation = (# samples with [Si] < 5 μM and [Si]/[DIN] < 1)/Total # Samples. All the data except 1992 are described by the line: Diatom Abundance = $(-1.01 \times 10^5) \times$ % Limitation + 8.67×10^6, $r^2 = 0.8990$.

Officer and Ryther (1980) originally proposed that increasing eutrophication would decrease the Si:N ratio and diatom abundance and enhance nondiatom growth. Smayda (1989, 1990) hypothesized that decreased Si inputs in relation to increasing inputs of other nutrients would stimulate harmful algal blooms (HABs). Although there are certainly many HAB species in this region (Dortch et al. 1997b; Dortch et al. in press; Parsons et al. in press; Dortch et al. accepted), the historical data are insufficient to assess whether their incidence has increased (Dortch et al. accepted). The exception is *Pseudo-nitzschia* spp., a group of toxic diatoms whose increased abundance over time has been linked to increasing nutrient inputs from the Mississippi River (Rabalais et al. 1996; Dortch et al. 1997b) (see Section 7.3).

Although the effect of Si limitation on species composition has been investigated because diatoms dominate the biomass and play a pivotal role in trophodynamics and carbon flux in this system (Turner et al. 1998), the effects of N and P limitation on species composition in this region have not been investigated. From studies in marine and freshwater systems elsewhere (e.g., Tilman et al. 1986; Makulla and Sommer 1993; Sommer 1989, 1993, 1994, 1995, 1996; Reigman 1992), N and P limitation will also strongly influence phytoplankton species composition, possibly in ways that will affect the function of the ecosystem (e.g., Reigman 1992).

6.5 CARBON FLUX

The production of carbon via photosynthesis in surface waters eventually contributes to the flux of carbon to the bottom and decline of oxygen through aerobic respiration. The fallout of surface material to bottom waters along the inner to middle continental shelf should be high for two reasons. First, primary production in these waters is high (290 to > 300 g C/m^2/yr) (Sklar and Turner 1981; Lohrenz et al. 1990). Second, Suess' (1980) empirically derived formula relating phytoplankton production in surface waters to the amount of material falling into sediment traps at depth predicts that 50% or more of the surface production reaches the bottom in depths equal to those of the hypoxic zone and that the percentage decreases as depth increases. The amount of phytoplankton biomass in the bottom waters across the Louisiana inner and middle continental shelf is high, often exceeding 30 µg/l, and a high percentage is composed of phaeopigments. This relationship holds for the hypoxic zone and for differences related to depth. The respiration rate is proportional to phytoplankton pigment concentration (Turner and Allen 1982b); thus, more oxygen is taken up where higher flux of materials reaches the lower water column and sediments.

Studies of particle flux on the Louisiana shelf are limited to floating (Redalje et al. 1994) and moored traps (Qureshi 1995). Redalje et al. (1994) deployed free-floating MULTITRAP systems in both river plume and adjacent shelf regions. They determined that vertical flux was highest in the plume (1.8 g C m^2/d) and the shelf (0.4 g C/m^2/d) during May 1992 and lowest in the plume (0.29 g C/m^2/d) and shelf (0.18 g C/m^2/d) during July–August 1990. Qureshi's traps were deployed on an instrument mooring at depths of 5 m and 15 m in a 20-m water column in spring, summer, and fall (station C6B, see Figure 2.1). They were baffled traps with an aspect ratio of 3:1. Collections were funneled into a tube with brine solution (45 ppt) and 1% glutaraldehyde, and were retrieved at one- to three-week intervals, depending on weather. Phytoplankton and fecal pellets were enumerated; carbon was estimated from volume calculations (verified with experimental data); and total C, H, and N were analyzed on an elemental analyzer.

The material that sank into the sediment traps was composed of fecal pellets, directly sinking phytoplankton, and other unidentified carbon (Figure 6.11). The unidentified carbon consisted of molts, dead zooplankton (it is impossible to distinguish between swimmers and dead zooplankton that fell into the traps), marine snow, and particles with adsorbed organic carbon. Carbon flux was high, i.e. approximately 500–600 mg C/m^2/d in 15-m water depth (Qureshi 1995; Redalje et al. 1994). The total carbon flux was much lower in 1992 than in 1991, and the seasonal pattern of the sources of carbon flux was also quite different between both years and depths. A large portion of the particulate organic carbon flux that reached the bottom was incorporated in zooplankton fecal pellets (55%; Qureshi 1995), but also as individual cells or in aggregates. Overall, the flux of fecal pellet carbon and other carbon exceeded that of phytoplankton direct-sinking carbon. Both phytoplankton and zooplankton carbon fluxes were a larger percentage in the spring and fall.

A rough estimate of the fraction of production exported from the surface waters (compared to seasonal primary production data of Sklar and Turner 1981) was highly variable and ranged from 10% to 200%, with higher percentages in spring (Qureshi 1995); estimates of Redalje et al. (1994) were also quite variable. The high particulate organic carbon flux was sufficient to fuel hypoxia in the bottom waters below the seasonal pycnocline (Qureshi 1995; Justić et al. 1996). The carbon fluxed via fecal pellets was usually sufficient to deplete the bottom-water oxygen reserve in the spring, thus creating hypoxic conditions that then prevailed through the stratified summer period. While the flux of organic material in summer contributes to hypoxia, it is inconsequential to the majority flux of particulates in the spring (Qureshi, 1995). Turner et al. (1998) combined the sediment trap fecal pellet carbon data, chlorophyll data, and seasonal and depth-related respiration rate experiments in an analysis of how these relationships varied around a Si:DIN ratio of 1:1, as delivered by the Mississippi River (lagged three months). They showed that (1) there is a strong vertical, rather than horizontal, coupling between oxygen consumption in bottom waters and organic loading from surface waters; and (2) higher water-column respiration rates are driven by river-derived nutrients stimulating *in situ* organic production that sinks to the bottom layers. They also showed that respiration rates in bottom waters were responsive to zooplankton fecal pellet production (as predicted by Qureshi 1995) and to diatom production. The trophodynamics of carbon production, zooplankton, carbon flux, and respiration were sensitive to the Si:DIN ratio in the riverine waters (see Section 6.8).

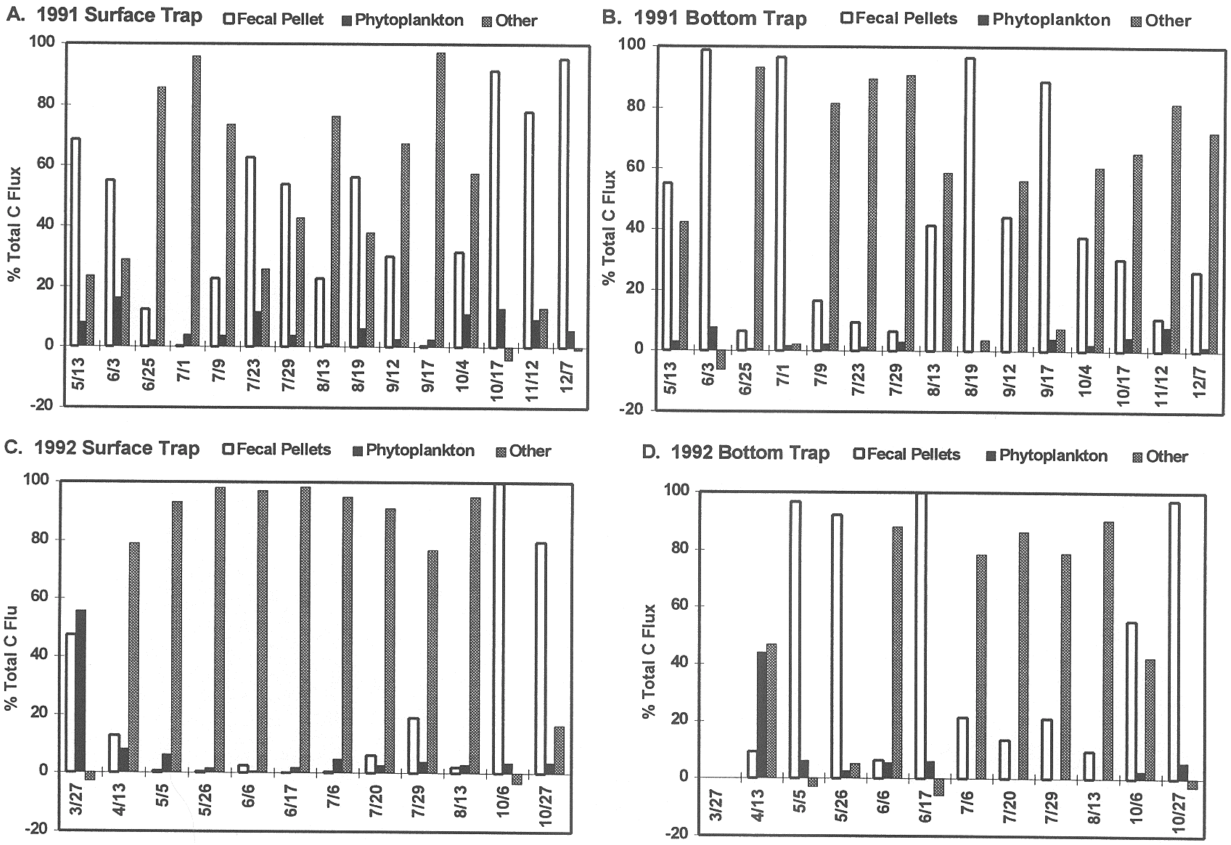


FIGURE 6.11. Percent total carbon flux comprised of fecal pellets, phytoplankton, and other material. NOTE: Fecal pellet carbon obtained from counts and volumes, using an empirically determined C/volume ratio (Qureshi 1995). Phytoplankton carbon calculated from chlorophyll using the C/Chl ratio for surface traps and bottom traps in 1991, calculated from counts and volumes of phytoplankton, converted to C. (*From Dortch 1994; Rabalais et al. 1998.*)

Because the sediment traps were serviced by divers, they were not deployed from late fall through early spring, when high fluxes of diatoms might have occurred (based on subsequent analysis of phytoplankton biomass and composition during that season). Our understanding of fluxes of carbon is currently limited to the spring through early fall period and to the two years of data that have been analyzed for the four years of trap collections. The differences in the two years of data indicate high interannual variability. One difference in the two years was the timing and discharge of the river. The flow in 1992 was much lower than in 1991. Correlations that existed between flux in 1991 with indicators of riverine discharge (low salinity, high chlorophyll *a* biomass) were not present in 1992.

6.6 COMPOSITION OF SINKING MATERIAL AND THE RELATIONSHIP TO NUTRIENT AVAILABILITY

Despite the limited carbon flux data, there were some clear links between different sources of carbon flux and nutrient availability in the coastal zone. In Figure 6.10 (and associated text) it was shown that both diatom abundance and the types of diatom species present were related to silicate availability in the water. Because of its effect on diatom abundance, silicate availability may also be a factor controlling phytoplankton and fecal pellet carbon flux. Diatoms comprised a much higher percentage of the total phytoplankton sinking into sediment traps than their percentage of the total phytoplankton in the surface water (Table 6.3), suggesting selective sinking of diatoms. The other phytoplankton found in sediment traps were primarily picocyanobacteria that did not contribute substantially to the carbon flux because of their small size.

TABLE 6.3. Abundance of diatoms relative to other phytoplankton in sediment traps moored in the hypoxic region's core (C6A and C6B) from March to September 1990 (11 deployments) and April to December 1991 (15 deployments) and in surface water.

	% Diatom/Total Phytoplankton	
	1990	***1991***
Top Trap	31	74
Bottom Trap	42	66
Surface Water	13	10

Because the peak in cyanobacterial flux was greater in spring when diatom abundance was greatest than it was summer and fall when cyanobacterial abundance was highest, it has been proposed that the cyanobacteria sink as part of diatom aggregates (Dortch et al. 1997a, 1998). The diatom species that did sink represent only part of the total diatom community, including heavily to moderately silicified species and excluding lightly silicified species that were present in the water column (Table 6.4) (Dortch et al. 1992b; Fahnenstiel et al. 1995; Dortch et al. 1997a, 1998). Thus, Si must be available for high sinking flux of heavily to moderately silicified diatoms. These results have an apparent contradiction. The greatest sinking fluxes occur when diatom abundances are high, which is also when Si limitation is much more likely to occur. Other data suggest that Si limitation triggers sinking in many diatom species (e.g., Bienfang et al. 1982).

TABLE 6.4. Abundance of diatoms relative to total phytoplankton and abundance of sinking diatom species relative to total diatoms in sediment traps moored in the core of the hypoxic region (C6A and C6B) from March to September 1990 (11 deployments) and in surface water.

	% Diatoms/ Total Cells	% Sinking Diatoms/ Total Diatoms
High Runoff		
Water	48	78
Top Trap	30	98
Bottom Trap	39	98
Low Runoff		
Water	1	45
Top Trap	32	99
Bottom Trap	44	99

Many zooplankton feed on diatoms, which are the most abundant food in the larger size range in this system. Fecal pellet production is often proportional to the availability of food (e.g., Corner et al. 1972; Butler and Dam 1994). In 1991 there was a close correspondence between fecal pellet flux and diatom abundance (Figure 6.12, left panel), although in 1992 there was not (Figure 6.12, right panel). With carbon flux data for only two years, one year cannot be labeled as unusual, but other data (Figure 6.10) suggest that 1992 may have been unusual. Because diatom abundance is dependent on silicate availability (Figure 6.10), Si availability may be a factor controlling fecal pellet production and flux; data from multiple years identified the link between Si availability, diatom abundance, and fecal pellet flux (Turner et al. 1998; see Section 6.8).

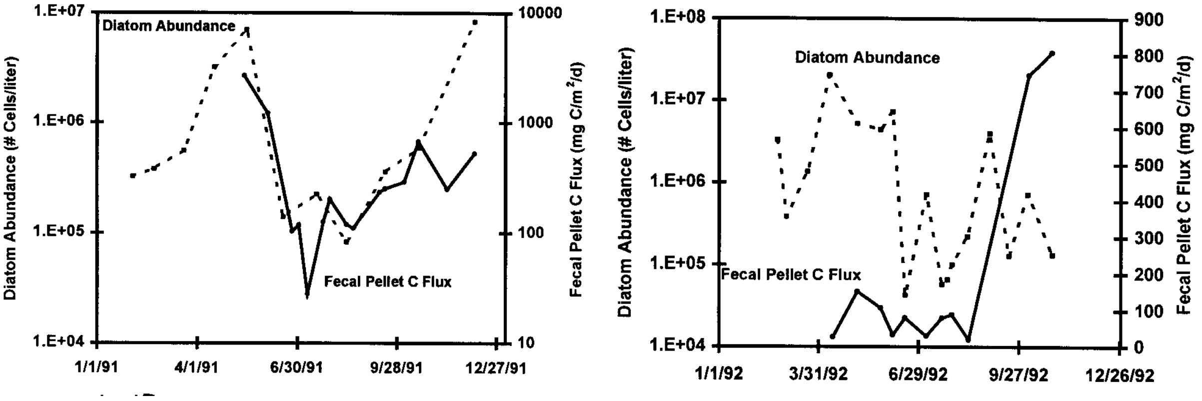


FIGURE 6.12. Diatom abundance and fecal pellet carbon flux at C6A and C6B in 1991 and 1992. *(From Qureshi 1995; Dortch unpublished data.)*

6.7 RESPIRATION

The oxygen consumption rates in near-bottom waters of the seasonally oxygen-deficient continental shelf were measured during several spring and summer cruises of multiple years (Turner et al. 1998; Turner and Rabalais 1998a). Rates varied between 0.0008 and 0.29 mg O_2/l/hr, and were sufficient to reduce the *in situ* oxygen concentration to zero in less than four weeks. The rates were inversely related to depth and decreased westward of the Mississippi River Delta, consistent with the decrease in nutrients and pigment concentrations. Respiration rates per unit phytoplankton pigment were highest in the spring, in shallower waters, and also closest to the Mississippi River Delta. These results indicate a strong vertical, rather than horizontal, coupling between oxygen consumption in bottom waters and organic loading from surface waters, and are consistent with the hypothesis that the higher water column respiration rates are driven by river-derived nutrients stimulating *in situ* organic production that sinks to the bottom layers.

Respiration in sediments is an additional oxygen sink for these waters. Rowe (in Dortch et al. 1994) suggested that this oxygen sink may sometimes equal respiration in the overlying waters, but most results (field and modeling experiments) indicate that the sediment consumption is seldom more than one-third of the total oxygen uptake below the pycnocline.

Dortch et al. (1994) suggested that photosynthesis on or near the sediment-water interface may occur and offset oxygen uptake processes at least to the point that anoxia does not occur more frequently. The low oxygen concentration observed in the samples for respiration experiments (average 3.4 mg/l, n = 40), however, suggested that benthic oxygen production was relatively low. Also, because of the high turbidity of the continental shelf waters near the Mississippi River, primary productivity is low below a depth of 10 m (Lohrenz et al. 1990).

Light conditions, however, could be an influential factor determining where hypoxic water masses were not located (Bierman et al. 1994). That is, the extinction coefficients might be improved sufficiently at the edge of the hypoxia water masses to affect oxygen production and consumption.

6.8 NUTRIENT RATIOS AND IMPLICATIONS TO BIOGEOCHEMICAL CYCLES AND TROPHODYNAMICS

The availability of dissolved silicate and its ratio to total inorganic nitrogen are important in controlling the extent of diatom production and the composition of the diatom community, with implications for carbon flux and control of oxygen depletion (Dortch and Whitledge 1992; Nelson and Dortch 1996; Turner et al. 1998). The lower concentration of silicate and a ratio of Si:N closer to the Redfield ratio could favor nonsiliceous forms of phytoplankton, such as dinoflagellates or cyanobacteria. On the other hand, it is plausible with the increase of N that larger, more heavily silicified diatoms that sink more readily and add to the oxygen demand of bottom waters would be competitively superior. Evidence supports both of these hypotheses in varying degrees (see discussion in Chapter 7 on historical changes in biological indicators).

Turner et al. (1998) demonstrated the coupling between oxygen consumption in bottom waters and vertical flux of carbon from nutrient-enhanced surface-water primary production. They also showed that carbon production, zooplankton, carbon flux, and respiration were sensitive to the Si:DIN ratio in the riverine waters. The percentage of copepods in the total mesozooplankton assemblage changed dramatically as the average Si:DIN ratio approached 1:1. Copepods made up about 30% of the mesozooplankton at a Si:DIN ratio of 0.5 and 75% at 1.0. Both the percentage of fecal pellet carbon in the top trap and the estimated primary production captured as fecal pellets were high when the average Si:N atomic ratio in the river was greater than 1:1. These two patterns were consistent with the expectation that copepod density and grazing rates would be higher when the production of the principal prey—diatoms—was higher. When the Si:DIN ratio was high, respiration rates declined with depth more slowly than when this ratio was low. This result is consistent with the idea that faster sinking rates will result in less organic matter consumption in transit to the bottom and a more uniform distribution of respiration rates in near-bottom waters of different depths.

Because the diatom-rich fecal pellets travel more quickly to the bottom than diatom-poor fecal pellets, the respiration rate per chlorophyll *a* accelerates with increasing Si:DIN ratio of the fresh water entering the system. The heavily silicified diatoms, produced mainly in high-discharge spring conditions, sink faster than diatoms with lightly silicified frustules produced in low-flow summer conditions, especially when packaged as fecal pellets. The proportion of primary production formed into fecal pellets, the water column respiration rate, and near-bottom respiration rates were sensitive to the Si:DIN ratio in the riverine waters, and changed dramatically near the Redfield ratio.

The patterns identified by Turner et al. (1998) were consistent with the hypothesis of Officer and Ryther (1980) that a shift in the Si:N atomic ratio from above 1:1 to below 1:1 would alter the marine food web by reducing the diatom-to-zooplankton-to-higher trophic level food web, and increasing the proportion of flagellated algae, including those that are potentially harmful. Turner et al. (1998) speculated on the implications of pivoting around the Si:N ratio of 1:1. The system appears to be poised to exhibit two very different food webs (Table 6.5). If flagellated algae become dominant, it does not mean that hypoxia will end. The blooms of flagellated algae, for example, may contribute to hypoxic water formation if sufficient biomass sinks from the surface to the bottom layer.

TABLE 6.5. Summary of observations and probable consequences with Si:DIN atomic ratios near 1:1 and less than 1:1.

Observations and Implications	**Si:DIN**	
	< 1:1	**1:1 or > 1:1**
Observations		
% of mesozooplankton that are copepods	Low	Dominant
% of carbon in sediment traps that is in fecal pellets	Low	High
% of primary production that is in fecal pellets	Low	Dominant
Sinking rate of surface carbon produced	Slower	Faster
Respiration per Chl *a* in bottom waters	Lower	Higher
Respiration losses in bottom waters	Lower	Higher
Implications		
Potential for flagellated algal blooms, including HABs	Higher	Lower
Bottom-water hypoxia zone	Less severe, more sporadic	Continuing severity

6.9 OXYGEN AND CARBON BUDGETS

Oxygen profiles and calculations of oxygen anomalies (Figure 6.13) for station C6* off Terrebonne Bay on the southeastern Louisiana shelf over the period 1985–92 were used to define oxygen and carbon budgets for an area that is consistently hypoxic on a seasonal basis (Justić et al. 1993, 1996, 1997). This area is suitable for the development of a coupled biological–physical two-box model for several reasons. Vertical oxygen transport is likely to be more important than horizontal oxygen transport for this area, because the data suggest a relatively high coherence between changes in vertical temperature gradients and changes in bottom oxygen concentration (Rabalais et al. 1992a) and a strong tidal signal of any kind, which would indicate horizontal transport is not present in the periodograms of oxygen data from station C6* (Rabalais et al. 1994). Also, maximum lateral displacement of only 3 km can be expected due to diurnal and semidiurnal currents (Rabalais et al. 1994).

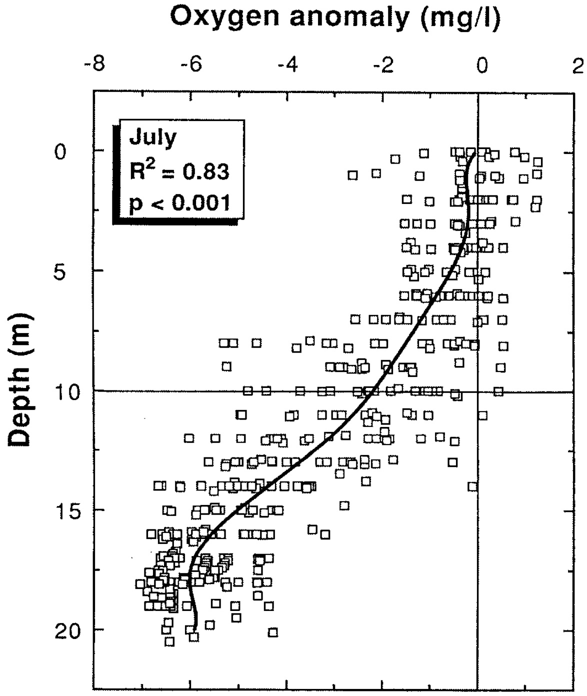


FIGURE 6.13. Oxygen anomaly (difference between measured oxygen and expected oxygen concentration at 100% saturation) for station C6* in 1985–92.

The surface-water layer, above the prevalent pycnocline at 10 m, shows an oxygen surplus during February–July, with the maximum in April and May, which coincides with mean peak Mississippi River flow (Figure 6.14). The bottom layer, below the pycnocline to 20 m, exhibits an oxygen deficit throughout the year (Figure 6.15), but reaches its highest value in July when surface-to-bottom density differences are greatest.

Based upon Figures 6.14 and 6.15, 90% of the annual net oxygen production at station C6* occurs between February and June (Justić et al. 1996, 1997) (Figure 6.16). The integrated annual net productivity (NP) of the upper water column (0–10 m) at station C6* is 423 g O_2/m^2. If an oxygen-to-carbon ratio of 3.47 by weight (mol O_2:mol C = 138:106; PQ = 1.3) is assumed for the photosynthetic process, then the total net carbon production is 122 g C m^2/yr. The excess of

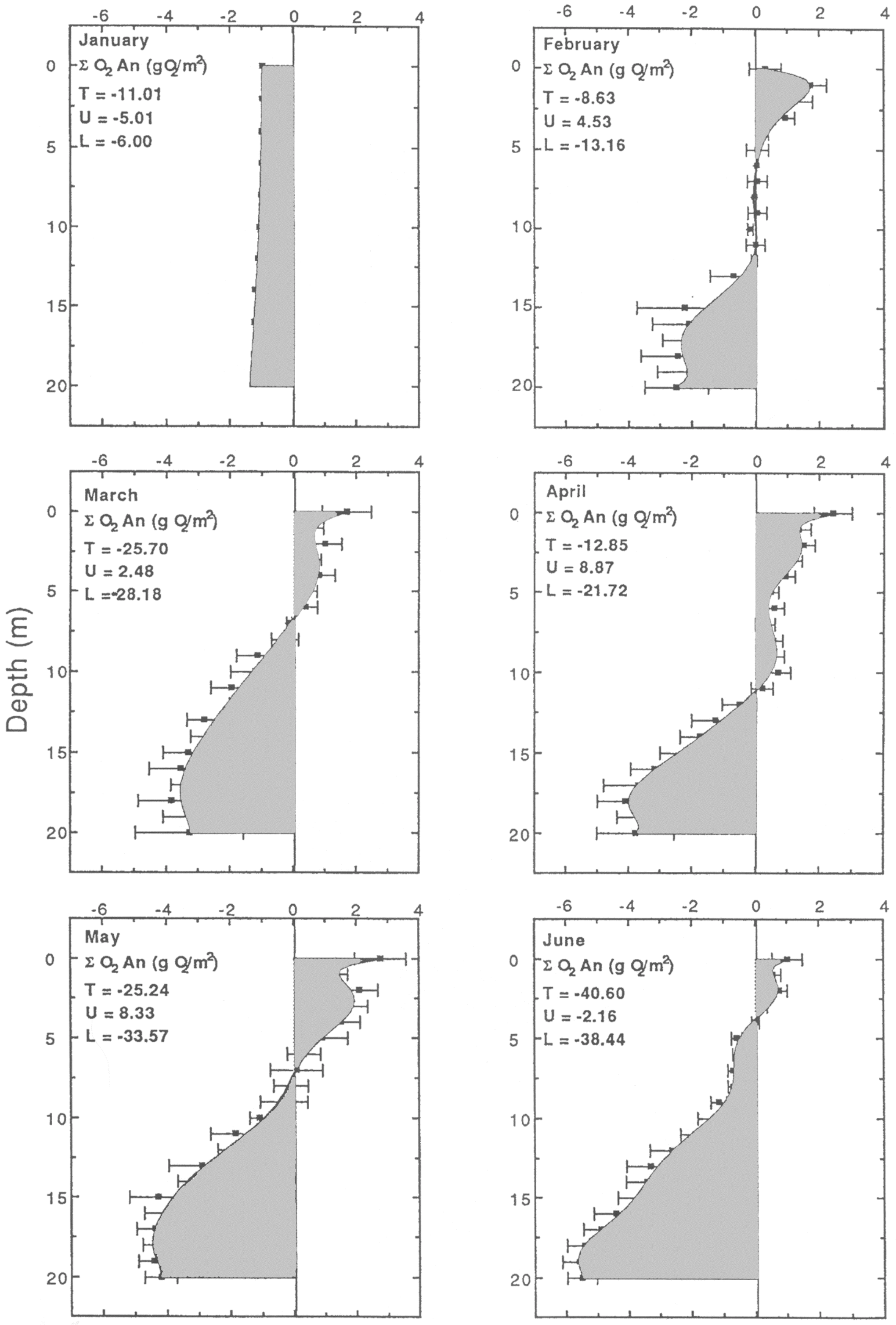


FIGURE 6.14. Mean monthly oxygen surplus or deficit at 1-meter intervals for station C6* for January–June 1985–92. *(From Justić et al. unpublished data.)*

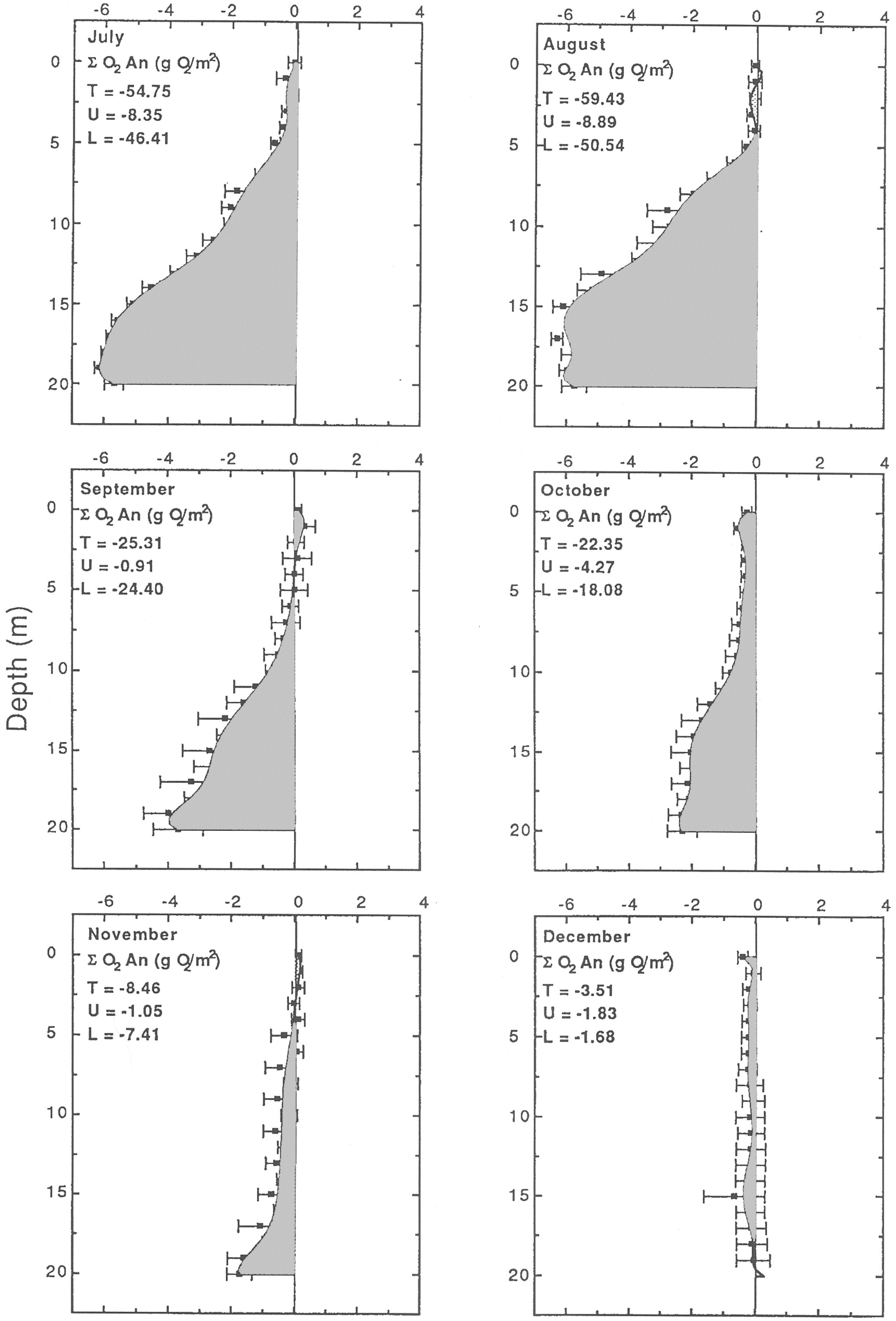


FIGURE 6.15. Mean monthly oxygen surplus or deficit at 1-meter intervals for station C6* for July–December 1985–92. *(From Justić et al. unpublished data.)*

organic matter, derived from primary production, is redistributed within the system and eventually decomposed in the lower water column and in the sediments. Integrated oxygen uptake rates for the lower water column (10-20 m) are significantly higher between January and June than the rest of the year. This correlates well with the seasonal changes in NP in the upper water column. The integrated annual oxygen uptake rate in the lower water column at station C6* is 197 g $O_2/m^2/yr$, which converts to a value of 57 g $C/m^2/yr$, if an RQ value of 0.77 (mol C:mol O_2 = 106:138) is used for the respiration process. Thus, on an annual basis, 47% of the surface net organic production at station C6* is decomposed in the lower water column and in the sediments (TR:NP = 0.47), a value that compares well with estimates of fixed carbon exported from surface waters to the lower water column (Qureshi 1995).

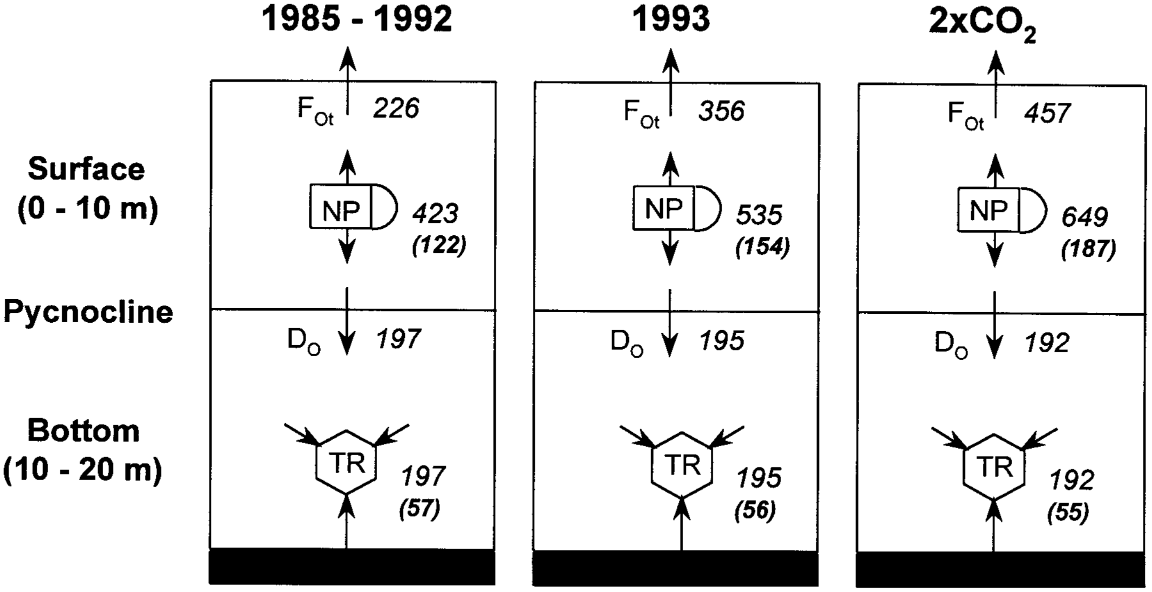


FIGURE 6.16. Global oxygen fluxes (g $O_2/m^2/d$) at Station C6* for the periods 1985–92 and 1993 and model projections for a doubled CO_2 climate. Note: F_{ot} denotes the total air–sea oxygen flux; NP is the net productivity of the upper water column (0–10 m); D_o is the diffusive oxygen flux through the pycnocline; and TR is the total oxygen uptake in the lower water column (10–20 m). Carbon equivalents, computed from the Redfield stoichiometric model (C:O_2 = 0.288, by weight), are given in parentheses. *(From Justić et al. 1997.)*

A high degree of coherence exists between the Mississippi River nitrate flux and net production rates at Station C6*. This allowed Justić et al. (1997) to predict that the net productivity of the upper water column has increased since the 1950s, coincident with increased river nitrate flux and concentration—a relationship verified with sediment cores (see Sections 7.4 and 7.5). If one assumes that the riverine nitrogen input was 50% lower than at present, the monthly nitrate flux during the 1950s did not exceed 1.6 x 10^6 kg/d. Based on the relationships of the model (Justić et al. 1996, 1997), that flux would be sufficient to support net productivity of about 0.29 g $C/m^2/d$, which is only 25% of the peak NP value for 1985–92 (1.15 g $C/m^2/d$). Consequently, the integrated annual net production was substantially lower than at present, and probably did not exceed 25 g $C/m^2/yr$. Even if losses due to export and burial were insignificant, and the total annual production of 35 g $C/m^2/yr$ was decomposed in the lower water column and sediments, the annual total oxygen uptake value (TR) would have been around 121 g $O_2/m^2/yr$. This result is substantially lower when compared with the estimates for 1985–92 (197 g $O_2/m^2/yr$) and 1993 (195 g $O_2/m^2/yr$). Thus, the net productivity of the upper water column appears to be an important factor controlling the accumulation of organic matter in coastal sediments and development of hypoxia in the lower water column.

The model for Station C6*, calibration data, experimental results from the 1993 flood, and doubled CO_2 climate scenario projections, as well as modifications to coastal carbon and oxygen budgets under anthropogenic nutrient enrichment throughout the world, indicate a close coupling between river-borne nutrients, net productivity, vertical carbon flux, and hypoxia on decadal time scales.

6.10 BIOGEOCHEMICAL PROCESSES OF LOW-OXYGEN ENVIRONMENTS

In general, the relationship between oxygen concentration and the concentration of dissolved inorganic nitrogen, phosphate, and silicate is inverse (Figure 6.17). At the lowest oxygen concentrations, the nutrient concentrations were generally higher. Nutrients are released during decomposition of organic matter that has settled into the lower water column from the surface. If dissolved oxygen is extremely low, N will remain in the more reduced state. Nitrification takes place until it becomes oxygen-limited as sediments approach anoxia.

For July 1993 LATEX data, there were elevated concentrations of NH_4^+, NO_3^-, NO_2^-, and PO_4^{3-} in hypoxic bottom waters, especially where oxygen levels were less than 0.5 mg/l (Rabalais and Turner 1998). Hypoxia was not as widespread and severe in July 1994 (LATEX data) as in 1993, but there were elevated levels of NH_4^+, NO_2^-, and PO_4^{3-} in water less than 0.5 mg/l oxygen. Most of the inorganic nitrogen is present as nitrate, but there were significant amounts of NH_4^+ and NO_2^- at the lowest oxygen concentrations, indicating organic decomposition in sediments and nutrient release (similar to hypoxic/anoxic environments in the mid-reaches of Chesapeake Bay (Cowan and Boynton 1996). Some of the NH_4^+ could have been converted to NO_3^- through nitrification.

Higher phosphate concentrations at the lower dissolved oxygen levels are consistent with a strong flux of P as sediments become anoxic (sulfide out-competes phosphate for iron-binding sites), as also seen in Chesapeake Bay. Higher dissolved silicate fluxes out of the sediments in hypoxic bottom waters in mid-Chesapeake Bay were related to the flux of organic matter from surface waters, but only after a temporal lag of one month was added between deposition events (shown as elevated sediment chlorophyll *a* concentrations) and sediment nutrient releases. One could predict then that those stations with higher SiO_4^- concentrations (Figure 6.17) received a greater flux of silicate-based organic material in prior months.

6.11 LINKAGES WITH MISSISSIPPI RIVER DISCHARGE AND NUTRIENT FLUX

Both the physical structure and the biological processes of the Louisiana continental shelf are influenced by the nutrient-rich freshwater discharge of the Mississippi River system. Freshwater discharge peaks in spring, as does nutrient flux, but the relationship of the two is not perfect. Physical differences in salinity from the surface to the bottom increase in the spring as well, and are strengthened in the summer with solar heating and reduced waves in calmer weather. There are time and spatial lags with nutrient delivery and production in the surface waters, and a subsequent lag in flux of carbon to the lower water column and oxygen uptake in the lower water column and sediments. Spatial and temporal variability in the distribution of hypoxia is related, at least partly, to the amplitude and phasing of the Mississippi River discharge and nutrient fluxes (Pokryfki and Randall 1987; Justić et al. 1993, 1996, 1997; Rabalais et al. 1996; Wiseman et al. 1997). A series of experiments, empirical relationships, seasonal oxygen and carbon budgets, time-series analyses, models, and comparisons with other regions of the world's coastal ocean indicate a close coupling between river-borne nutrients, net productivity, vertical carbon flux, and hypoxia on short (day) to long (decadal to century) time scales (Table 6.6).

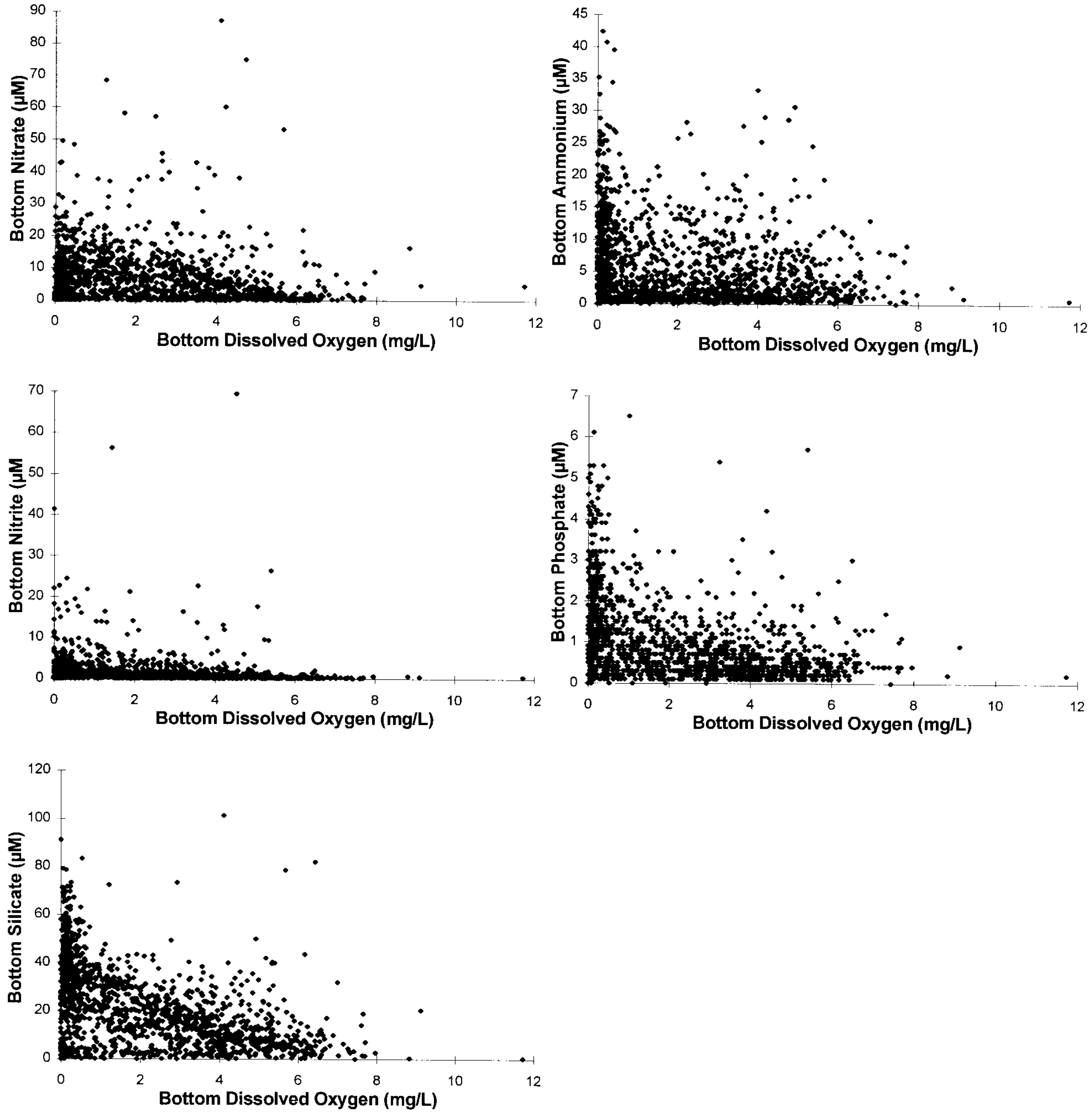


FIGURE 6.17. Comparison of bottom-water dissolved oxygen concentration and bottom-water dissolved inorganic nutrient concentrations for all stations and all depths for May–September 1985–96. (*From Rabalais and Turner unpublished data.)*

TABLE 6.6. Evidence for nitrogen-driven phytoplankton production.

Temporal Scale	Evidence
Days	Bioassay experiments. Simulated *in situ* measurements of primary production across a range of dissolved inorganic nitrogen concentrations. Short-term primary production models. Correlation of N-NO_3 with primary production. Depletion of NO_3 and Si along a salinity dilution gradient. *In situ* Redfield ratios.
Months	Correlation of primary production w/ time-lagged nutrient concentration & flux. Correlation of surface-water net production with time-lagged nutrient flux. Correlation of surface-water net production and bottom-water oxygen stress with 1-month and 2-month, respectively, lagged freshwater discharge. Response of mass-balance model to reductions in nitrogen load. Oxygen and carbon budgets. Carbon flux relationships with indicators of river discharge and surface-water increased production.
Years	Sediment cores and coincidental timing with increased nitrogen loading. Increase in accumulation of increased marine-source carbon. Increase in silicate-based productivity. Increase in foraminiferan index of A/E (increased carbon accumulation and worsening oxygen stress). 1998, 1992, or other low-discharge years for nonevents; variability in spring discharge, predicted C flux, or stratification, or combination.

Net productivity (a surrogate for excess carbon available for export) of the upper water column appears to be an important factor controlling the accumulation of organic matter in coastal sediments and development of hypoxia in the lower water column. Seasonal dynamics of net productivity in the northern Gulf of Mexico are coherent with the dynamics of freshwater discharge (Justić et al. 1993). The surface layer (0–0.5 m at station C6*) shows an oxygen surplus relative to the saturation values during February–July; the maximum occurs during April and May and coincides with the maximum flow of the Mississippi River (Figure 6.18). In contrast, the bottom layer (approximately 20 m) exhibits an oxygen deficit throughout the year, but reaches its highest value in July. Bottom hypoxia in the northern Gulf is most pronounced during periods of high water column stability when surface-to-bottom density differences are greatest (Section 4.2) (Rabalais et al. 1991; Wiseman et al. 1997). The correlation between Mississippi River flow and surface oxygen surplus peaks at a time lag of one month, and the highest correlation for bottom oxygen deficit is for a time lag of two months (Justić et al. 1993) (Figure 6.18). These findings suggest that the oxygen surplus in the surface layer following high flow depends on nutrients ultimately coming from the river but regenerated many times.

Annual mass-balance calculations (Turner and Rabalais 1991; Dortch et al. 1992a) and N uptake measurements in the fall suggest that N atoms are recycled on the average about four times, although recycling may be less important in the spring (Dortch et al. 1992a; Bode and Dortch 1996). This finding is important, because a surplus of oxygen relative to the saturation value is a good indicator of net productivity in the surface waters. An oxygen surplus also means that there

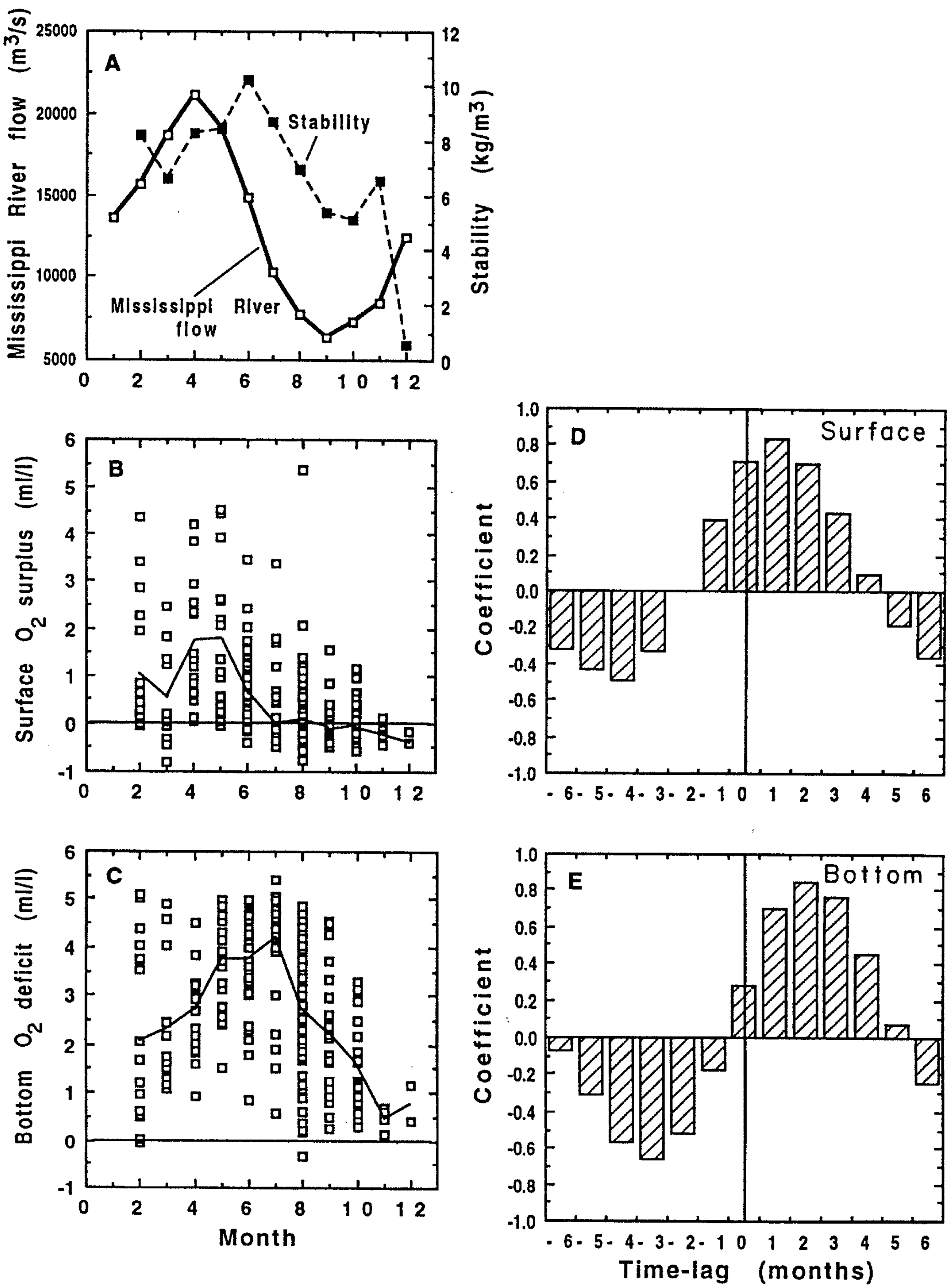
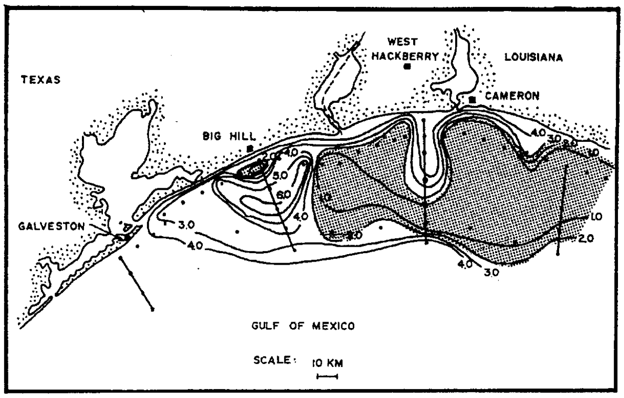


FIGURE 6.18. Left panel: Seasonal changes in river flow, stability, surface oxygen surplus, and bottom oxygen deficit for station C6*. NOTE: The solid lines represent the monthly means for 1985–91. Surface-oxygen surplus and bottom-oxygen deficit are O_2- O_2' and –(O_2- O_2'), respectively, where the O_2 was the measured oxygen concentration and O_2' was the oxygen concentration at 100% saturation. **Right panel: Cross-correlation analysis of the data on river flow versus surface oxygen surplus and bottom oxygen deficit.** (*Modified from Justić et al. 1993.)*

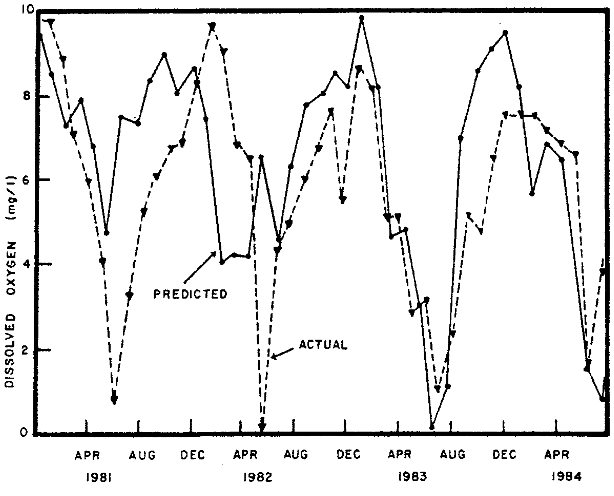
is an excess of organic matter derived from primary production that can be redistributed within the system; much of this will eventually reach the lower water column and sediments. The development of summer hypoxia in the northern Gulf of Mexico is associated with the decay of organic matter accumulated during spring phytoplankton blooms (Qureshi 1995) (see Sections 6.5 and 6.6 on carbon flux). These findings demonstrate a close coupling between river-borne nutrients, net productivity, and hypoxia, as well as implicate the effects of anthropogenic nutrient loads on a coastal marine ecosystem.

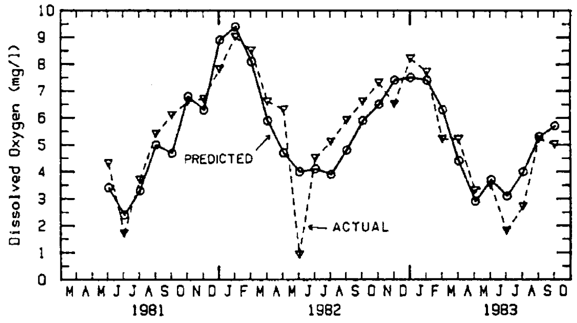
Nutrients clearly stimulate primary production in the Mississippi River Bight region (Lohrenz et al. 1997). The relationships between dissolved inorganic nitrogen flux and concentration and orthophosphate concentration were strongest for the eastern and central parts of the bight. The relationship was equally strong within the western region of the study area, but the strength of the relationship improved with a lag in the delivery of the nutrients from the river delta. Justić et al. (1997) demonstrated a strong relationship between dissolved inorganic nitrogen flux and net production in the surface waters of station C6* (the western region of the Lohrenz et al. 1997 study), with a stronger relationship at a one-month lag for river flux. These two separate analyses clearly show the enhancement of primary production and net production with nutrient (N and P) increases.

Similar relationships with freshwater discharge and oxygen depletion in bottom waters at stations in a location west of the Atchafalaya River Delta and expected direction of materials and freshwater flux were identified by Pokryfki and Randall (1987) (Figure 6.19). Time lags were apparent between values of river discharge, bottom dissolved oxygen, and salinity. The highest cross-correlation coefficient between bottom-water dissolved oxygen (in the area off the Calcasieu Estuary) and river discharge (from the Atchafalaya) was –0.51 at a lag of two months. These results are similar to those of Justić et al. (1993) for station C6* on the southeastern shelf. The linear-regression model developed by Pokryfki and Randall (1987) did not include any factors for biological processes, and the accuracy would have been improved by “incorporating a biological component into the time series.”



Bottom contours of dissolved oxygen values (mg/liter) for the Northwestern Gulf of Mexico on July 9-10, 1984





Maximum cross correlation between bottom dissolved oxygen and river discharge was -0.51 at a lag of two months.

FIGURE 6.19. Upper panel: Location of bottom-water hypoxia on the southwestern Louisiana shelf in July 1984. Left panel: Comparison of actual and predicted bottom dissolved oxygen values from the linear regression model. Right panel: Comparison of actual and predicted bottom dissolved oxygen values from the time series model. (*From Pokryfki and Randall 1987.*)

CHAPTER 7

Long-Term Coastal Ecosystem Changes

Given the high volume of fresh water delivered by the Mississippi River and associated nutrients and the stratified coastal system, one might expect a propensity for the ecosystem to develop hypoxia naturally. In other words, has hypoxia always been a feature of the northern Gulf of Mexico adjacent to the discharge of the Mississippi River?

There is a general consensus that the eutrophication of estuaries and enclosed coastal seas worldwide has increased over the last several decades. Evidence from many coastal seas suggests a long-term increase in frequency of phytoplankton blooms, including noxious forms (Smayda 1990; Hallegraeff 1993). Also, an increase in the areal extent and/or severity of hypoxia was observed, for example, in Chesapeake Bay (Officer et al. 1984), the northern Adriatic Sea (Justić et al. 1987), some areas of the Baltic Sea (e.g., Andersson and Rydberg 1987), and many other areas in the world's coastal ocean (Diaz and Rosenberg 1995). Therefore, given the magnitude of changes in the nutrient flux to the Gulf, it would not be surprising that the northern Gulf of Mexico has experienced eutrophication and ecosystem-level responses to increased nutrients, changing nutrient ratios, and carbon flux. Unfortunately, the long-term data sets that demonstrate changes in surface-water productivity and bottom-water dissolved oxygen, such as those available for the northern Adriatic Sea and areas of the Baltic and northwestern European coast, do not exist for the northern Gulf of Mexico. Data from oceanographic cruises prior to 1985 with which to test these hypotheses are limited. Other sources of information, therefore, must be sought that record whether changes in marine ecosystem-level indicators have occurred as Mississippi River nutrients have changed.

7.1 HISTORICAL DOCUMENTATION

7.1.1 Oxygen Minimum Layer

The oxygen minimum layer is a permanent feature of the open Gulf of Mexico at 400–700-m depths. The mention of low oxygen conditions from the northern Gulf of Mexico for the mid-1930s can be traced to Conseil Permanent International pour l'Exploration de la Mer (1936) *Bulletin Hydrographique* for 1935. These conditions were also identified in Hedgpeth's *Treatise on Marine Ecology and Paleoecology* (Brongersma–Sanders 1957; Richards 1957) as records from the oxygen minimum layer. The data from the M/V *Atlantis* surveys for the Gulf of Mexico in 1935 (all stations north of 27°N indicated in Figure 7.1 top panel, including some shelf stations) were reviewed, and those with characteristic mid-water oxygen minimum zone measurements were plotted (bottom panel of Figure 7.1) and compared to present-day shelf hypoxia (shaded area represents 1996 bottom hypoxia distribution).

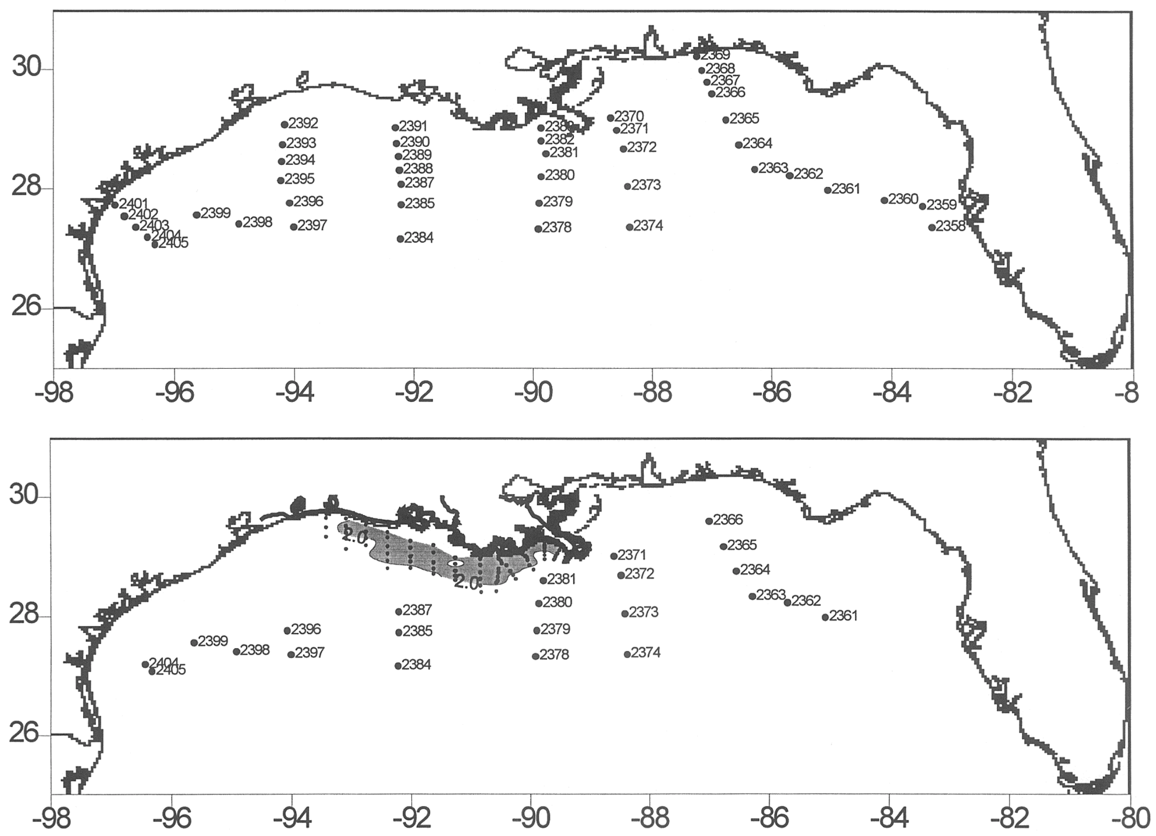


FIGURE 7.1. Upper panel: Stations from the 1935 M/V *Atlantis* surveys of the Gulf of Mexico north of 27°N. Bottom panel: M/V *Atlantis* stations with mid-water dissolved oxygen concentrations less than 3 cm^3/l (N.T.P.) (= 4.3 mg/l), compared to the hypoxic zone of July 1996. (*M/V* Atlantis *data from Conseil Permanent International pour l'Exploration de la Mer (1936); July 1996 data from Rabalais et al. unpublished data.*)

The stations with mid-water oxygen concentrations less than 3 cm^3/l (= 4.3 mg/l) were used as a very conservative measure of the occurrence of low-oxygen conditions. There were no values of oxygen less than 2 cm^3/l (= 2.8 mg/l) or less than 1.4 cm^3/l (= 2 mg/l), which more closely approximate the hypoxic conditions of the Louisiana continental shelf. Further, oxygen values less than 3 cm^3/l (= 4.3 mg/l) were well off the continental shelf and in water depths of 250–600 m of a total water column that was usually much deeper. Oxygen concentrations for *Atlantis* stations less than 50-m deep on the northern Gulf shelf (measured in February and March) were all above 5 cm^3/l. The months of February and March are less likely to be hypoxic, but not exclusively, because values below 2–3 mg/l have been recorded in these months (Rabalais et al. unpublished data). Thus, the idea that low-oxygen conditions have been reported from the Gulf since the 1930s is clearly erroneous. The oxygen minimum layer in deeper waters is disjunct from the continental shelf hypoxia that has been documented since the 1970s (Figure 7.1).

Earlier researchers cited in Turner and Allen (1982a) proposed that intrusion of the oxygen minimum layer from deeper waters onto the continental shelf, especially in the area of the Mississippi Canyon, was the source of bottom water depleted of oxygen. Several authors have shown that the oxygen minimum zone has no physical connection with the hypoxia on the inner to mid-continental shelf (Pokryfki and Randall 1987; Rabalais et al. 1991). Zones of hypoxic bottom waters are confined to the continental shelf, terminate well inshore of the shelf break, and are not continuous with the oxygen minimum layer of the Gulf of

Mexico (see above). In addition, the salinity, temperature, and respiration rates of water in the deep-water oxygen minimum zone in the Gulf of Mexico differ considerably from the continental shelf hypoxic waters. The oxygen consumption rates in the oxygen minimum layer are insufficient (by several orders of magnitude) to account for the observed decline in oxygen concentrations on the shelf (Turner et al. 1997).

7.1.2 Early Reports of Louisiana Shelf Hypoxia

Continental shelf hypoxia was first reported in the northern Gulf of Mexico in the early 1970s off Barataria and Terrebonne/Timbalier Bays as part of environmental assessments of oil production and transportation development studies (the Offshore Ecology Investigation (OEI), Ward et al. 1979; the Louisiana Offshore Oil Port (LOOP), Turner et al. in review) (Figure 7.2). The OEI study was conducted in Timbalier Bay and on the southeastern Louisiana shelf near the present-day stations of transects B and C of the hypoxia studies of Rabalais, Turner, and Wiseman (Figure 2.1). Severely depleted bottom waters were documented in August 1972 and July 1973 at stations in water 10–20 m deep. From May 1973 to May 1974, hypoxia was widespread between Barataria and Timbalier Passes in water 6–33 m deep and extended up to 30 km offshore. The years 1973 and 1974 marked consecutive 100-year floods of the Mississippi.

Following the initial documentation of hypoxia in 1972–74, Ragan et al. (1978) and Turner and Allen (1982a) conducted shelf-wide surveys and surveys in 1975 and 1976 between Mobile Bay and Atchafalaya Bay, respectively. The 1975 and 1976 distribution maps were less extensive than those mapped since 1985 by Rabalais et al. (1991, 1998), but the study area of Turner and Allen (1982a) at least was smaller than the current 60- to 80-station grid of the Rabalais et al. hypoxia studies. Turner and Allen (1982a) documented hypoxia (1) in March 1975 off Terrebonne Bay near the present-day Station C3, (2) in July 1975 over a fairly large area off Barataria and Terrebonne Bays, and (3) in July 1976 at one station off Terrebonne Bay near the present-day station of C6 (Figure 7.1). Values of oxygen in the Turner and Allen (1982a) survey were not as low as those documented during studies conducted in 1973–74. Ragan et al. (1978) surveyed the Louisiana shelf in 1975–76 and documented hypoxia in September 1975, July 1975, and August 1976 in the Mississippi River Bight and west of the Atchafalaya outflow (Figure 7.2).

The Central Gulf Platform Study was conducted at selected oil and gas production platforms and “control” areas between the Mississippi and Atchafalaya Deltas during April 1978–January 1979 (Bedinger et al. 1981). Stations were located in water depths of 10–200 m. Hydrographic measurements were taken at the surface and at 10-m intervals, so that the deepest measurements often did not reach near the bottom. Hypoxia was documented at several stations up to 27 m deep and was assumed to be present at others when benthic fauna were depauperate (June, July, and August 1978). A directed survey of bottom-water oxygen conditions conducted in August–September of 1978 outlined a moderately large area of hypoxia on the southeastern Louisiana shelf from the Mississippi River Bight to the Isles Dernieres and isolated areas off Atchafalaya Bay (Figure 7.2). The discharge of the Mississippi River was above average in 1978.

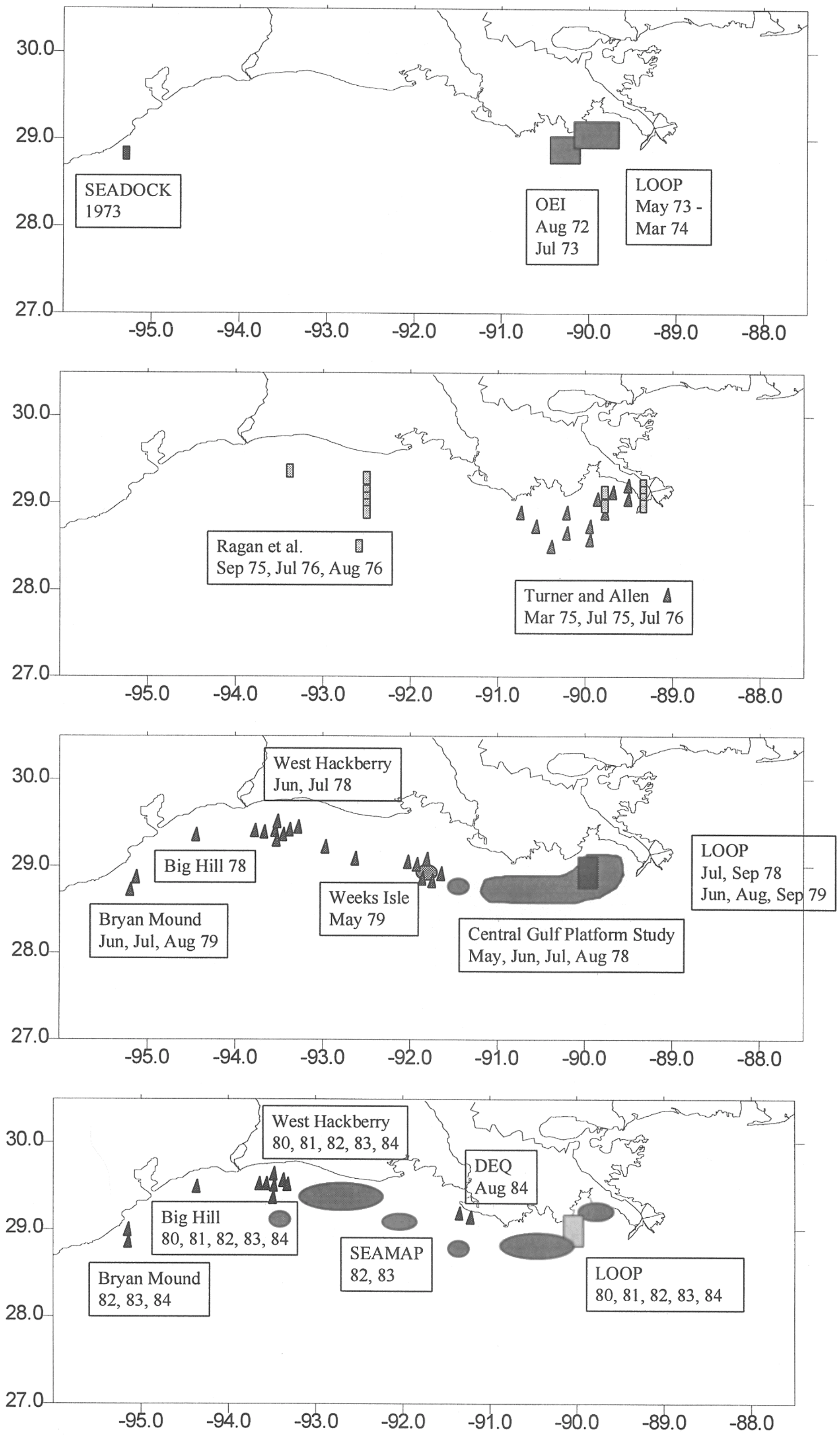


FIGURE 7.2. **Summary of studies, locations, and dates for which hypoxia was recorded.**

During 1978–84 several studies were conducted as part of the Strategic Petroleum Reserve Program to investigate the environmental conditions at several offshore locations for disposal of brine solution. These included the 1978–79 Texoma and Capline salt dome surveys off Louisiana (Jackson and Faw 1980) and the 1980–82 West Hackberry and Big Hill brine disposal sites off southwest Louisiana and the upper Texas coast (Harper et al. 1981, 1991; Kelly et al. 1983, 1984; Gaston 1985; Gaston et al. 1985; Pokryfki and Randall 1987). The locations of these sites are marked by triangles in Figure 7.2, along with the dates during which bottom-water hypoxia was recorded. Occurrences were in the summer months and were usually confined to a thin bottom layer.

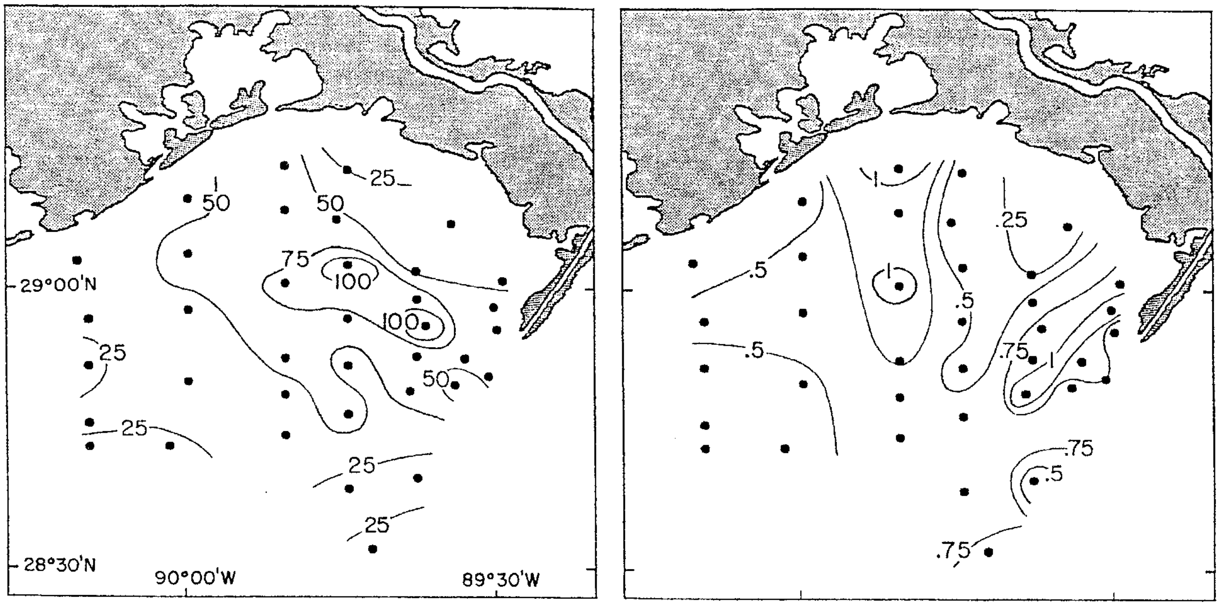
The National Marine Fisheries Service SEAMAP (Southeast Area Monitoring and Assessment Program) cruises that began in 1982 provide some data concerning the distribution of hypoxia on the Louisiana and Texas shelves (Gulf States Marine Fisheries Commission 1982 et seq.). The locations of hypoxia in the cruises of 1982 (first week of July) and 1983 (June 30–July 6) are shown in Figure 7.3 (based on data in Stuntz et al. 1982; Leming and Stuntz 1984; Renaud 1986). A preliminary analysis of SEAMAP data (K. Craig, unpublished data) documented hypoxia in isolated locations on the Texas and Louisiana shelves in 1983 and 1984, and more extensive areas in 1985 and in 1991–96.

Documented shelf hypoxia dates back to 1972. Prior to the 1970s, there is anecdotal information from shrimp trawlers in the 1950s–60s of low or no catches, of "dead" or "red" water, with the assumption that the lack of catch was related to low oxygen. Low catches, however, may not be due to low oxygen. There is no systematic examination of these anecdotal data. The tendency has been to generalize that low-oxygen conditions have always been a feature of the system; however, measurements do not exist to substantiate this statement. Surrogates for changes in the ecosystem, either water column or benthic habitat, can be derived from analyses of the sedimentary record and other data sources; these surrogates show eutrophication of the shelf and an increase in oxygen stress.

7.2 DATA AVAILABILITY

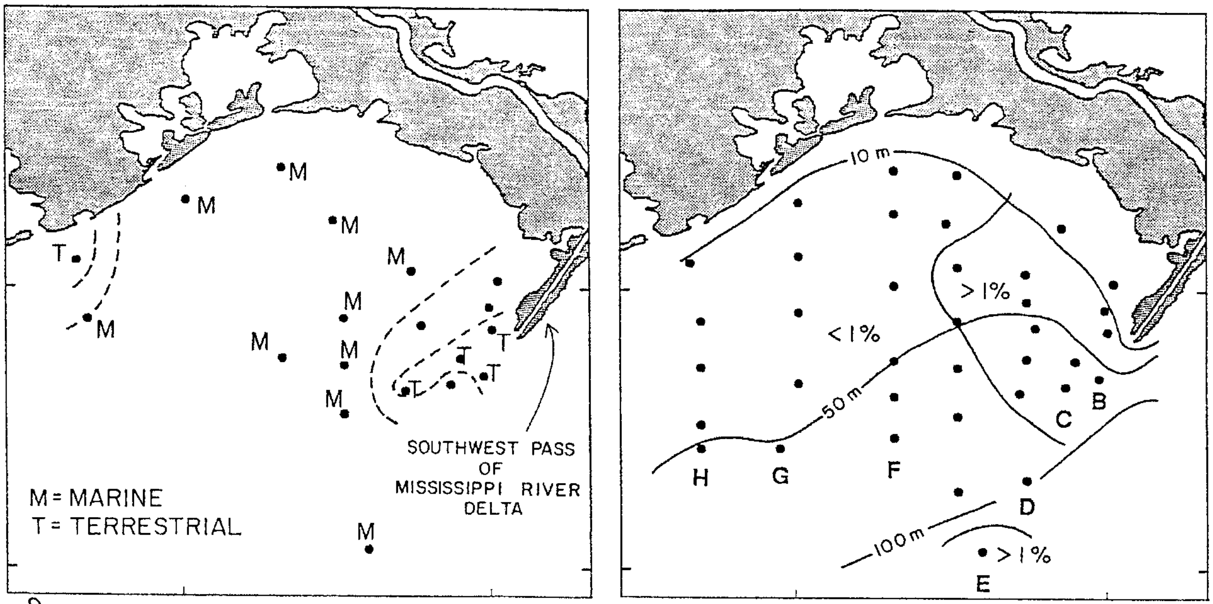
Long-term changes in productivity or changes in the severity and extent of hypoxia cannot be assessed directly from hydrographic data, because historic hydrographic data sets are limited, and systematic sampling of bottom-water dissolved oxygen concentrations did not begin until 1985. A few data sets from the 1950s to the present are suitable for comparisons of selected parameters. In addition, biological, mineral, or chemical indicators of surface-water production and hypoxia preserved in accumulating sediments record historical changes and provide clues to prior hydrographic and biological conditions. These many, disparate data sources provide a description of ecosystem-level changes that have occurred in the northern Gulf of Mexico within the framework of water quality changes in the Mississippi River.

Sediments accumulating under the Mississippi River plume provide historical information not available from hydrographic data or water samples. Surficial sediments, directly downstream and beneath the surface riverine–Gulf dilution plume reflect the *in situ* primary production and subsequent transport of organic carbon from surface to bottom waters within the Mississippi River Bight (Rabalais et al. 1992b; Turner and Rabalais 1994b) (Figure 7.3). The abundance of diatoms peaks at intermediate salinities down-current from fresh waters delivered from Southwest Pass and at intermediate distances from the pass (Turner unpublished data). The distribution of diatoms in the surface waters parallels the distribution of phytoplankton pigments and remnants of diatoms (biogenic silica) in the sediments beneath these surface features. These sedimentary habitats also represent a range of conditions relative to oxygen stress, from low frequency of occurrence to high frequency in mid-summer surveys (Sen Gupta et al. 1996) (Figure 7.4).



SEDIMENT TOTAL PIGMENTS (μg/g dry weight)

SEDIMENT BIOGENIC SILICA (weight %)



$\delta^{13}C$

SEDIMENT TOTAL ORGANIC CARBON

FIGURE 7.3. Spatial distribution in April 1989 of total phytoplankton pigments, biologically bound silica (BSi), δ^{13}C signature, and total organic carbon in sediments beneath the Mississippi River Delta plume. NOTE: Transect letters are identified in the lower right panel. There are seasonal peaks in pigments and BSi in spring and decreases in summer. (*From Rabalais et al. 1992b; Turner and Rabalais 1994b.*)

Sediment accumulation rates within 50 km of the Mississippi River Delta are 0.5–2.0 cm/yr (transects B–G in Figure 7.3) and generally < 0.3 cm/yr farther away (Turner and Rabalais 1994b). The best detail from dated sediment cores, therefore, is gained from data collected at intermediate depths (27–50 m). Differences seen in cores from deeper water and from greater distance from the delta, however, demonstrate the relative importance of various factors in those environs. The information in dated sediment cores documents changes in the area of the Mississippi River Bight under the direct influence of the discharge of the Mississippi River. Similar effects are expected in the influence of the plume of the Atchafalaya River, with suspected differences related to changes in discharge over time and/or differences in nutrient chemistry; however, these data do not exist nor are they likely to be acquired because sediments do not accumulate there.

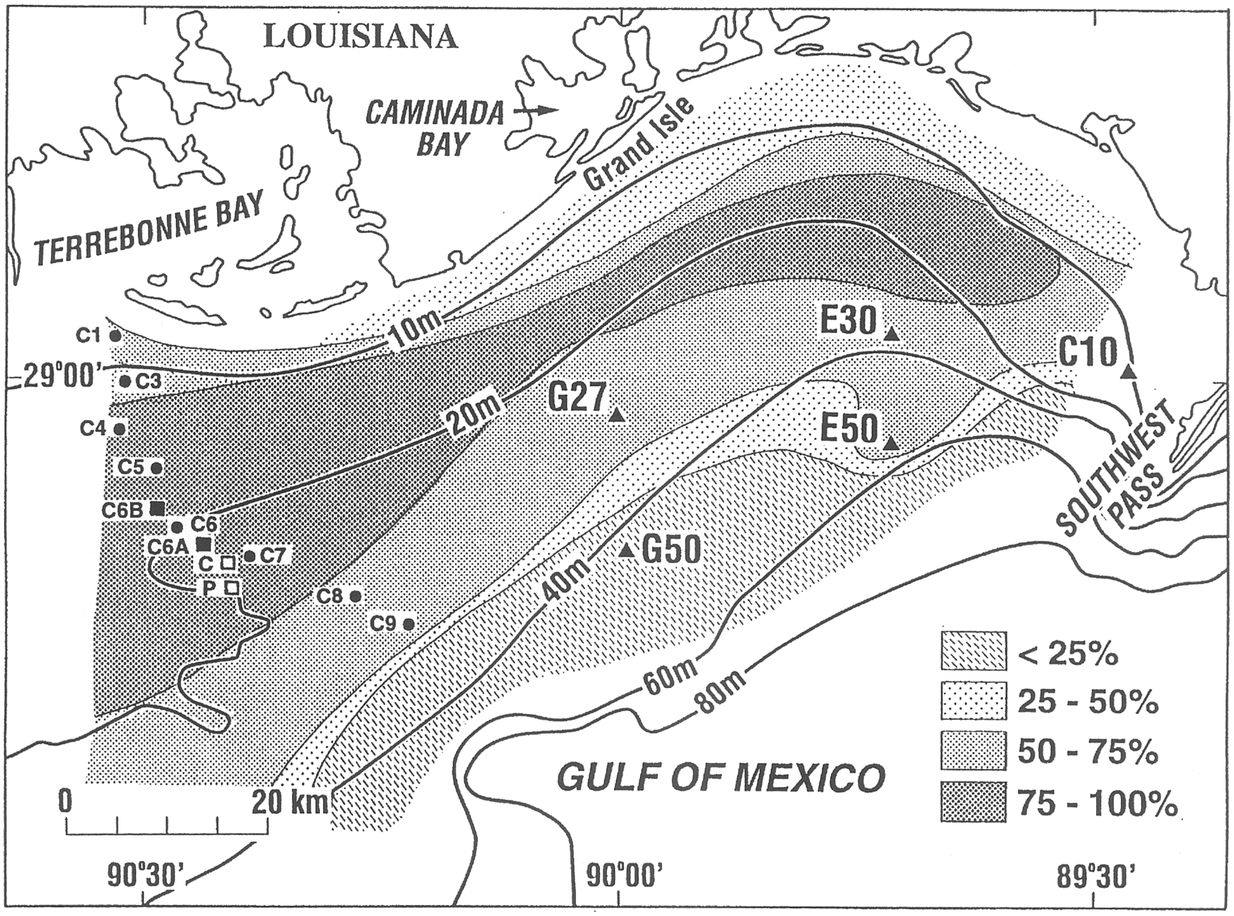


FIGURE 7.4. Station locations within the Mississippi Bight for transect C, mooring locations (C6A and C6B), sediment core stations (closed triangles), and "Platform" and "Control" stations of Fucik (1974) and Ward et al. (1979) (open squares marked with "C" and "P"). NOTE: Stippled intensity corresponds to frequency of occurrence of mid-summer hypoxia (1985–87, 1990–83). Station 10 of Eadie et al. (1994) and Nelsen et al. (1994) is close to E30. (*Modified from Rabalais et al. 1996.*)

7.3 CHANGES IN PHYTOPLANKTON COMMUNITY COMPOSITION

The changes in riverine and coastal nutrient concentrations and ratios over time suggest that changes in phytoplankton species composition should be observable. The increasing N availability and decreasing Si:N ratios should lead to increases in dominance of lightly silicified diatoms and nondiatoms.

Published reports of phytoplankton species composition for 1955–57 near the delta (Simmons and Thomas 1962) and for 1972–73 approximately 80 km west of the delta (Fucik 1974; Ward et al. 1979) were compared with recent data (1990–93) from near the delta and at stations C6A and C6B in water 20 m deep off Terrebonne Bay (Dortch et al. unpublished data compiled in Rabalais et al. 1996) (Figure 7.4). This comparison was qualitative because of differences in locations, seasons sampled, and methodology. The methodology used in earlier reports may have missed common nondiatoms, such as small coccoid cyanobacteria and phytoflagellates that now often dominate. Consequently, it was not possible to determine whether the dominance of these groups has increased. A conservative approach was taken in this comparative analysis. Data were compiled by matching season and location as closely as possible and by including in the recent data only phytoplankton types that would have been observed in the earlier studies. In addition, because of uncertainties of taxonomy, differences were only noted where organisms were identified unambiguously.

Demonstrable changes have occurred in the diatom and nondiatom species composition from the 1950s and 1970s to the present. Some heavily silicified diatom species are either not observed at all in recent samples or are much less dominant. For example, no *Melosira* species, which appear heavily silicified, were observed in 1990–93, but were present in both 1955–57 and 1972–73. *Asterionella japonica* (= *Asterionellopsis glacialis*, Round et al. 1990) was observed at low salinities in spring of 1990–93, but was not a dominant species as it was in 1955–57.

Similarly, more lightly silicified diatoms were documented for the 1970s and in 1990–93, especially at higher salinities. *Rhizosolenia fragilissima* and *Ceratulina pelagica*, which are so lightly silicified they are sometimes difficult to see, were not reported for 1955–57. Two species of *Leptocylindricus* spp. were frequently dominant in 1990–93, but were a minor constituent in 1955–57. During 1972–73 lightly silicified diatoms were reported, including *Rhizosolenia delicatula*, *Leptocylindricus danicus,* and *Ceratulina pelagica,* but a more quantitative analysis would be required to determine if their abundance was less than at present.

The data suggest that a shift in dominant diatom composition toward less silicified species occurred between 1955–57 and 1972–73, but methodological differences preclude conclusions about changes in nondiatoms. Because Si availability continued to decrease after the early 1970s (Turner and Rabalais 1991), a continued shift in species composition to nondiatoms would be expected. The phytoplankton at C6A and C6B in 1990–93 were often numerically dominated by small flagellates and cyanobacteria. They were not considered in this comparison, because it was not clear whether they would have been observed in the 1972–73 study, even if present in large numbers. It is also tempting to hypothesize that the presence of *Trichodesmium* sp. in 1990–93, but not in the earlier studies, indicates decreased Si availability; however, this bloom-forming species has been previously reported for this region (Eleuterius et al. 1981).

Several species important to human health are now present, but were either absent in the 1950s and 1970s samples or have increased in dominance. The dominance of *Pseudo-nitzschia* spp. on the Louisiana shelf appears to have increased dramatically since the 1950s, and concentrations now frequently exceed 1 x 10^6 cells/l. Some forms of this species have been associated with amnesiac shellfish poisoning (reviewed in Shumway 1990; Hallegraeff 1993). Because this species is heavily silicified for a marine diatom (Conley et al. 1989), it is difficult to attribute its increase to decreasing Si:N ratios, although the species could be responding to increasing N availability (Dortch et al. 1997a). *Dinophysis caudata*, a dinoflagellate that may be associated with diarrhetic shellfish poisoning (Dickey et al. 1992), was not recorded in the earlier studies. It was often present in the 1990–93 samples at low to moderate concentrations, but was sometimes dominant and reached concentrations as high as 1 x 10^5 cells/l.

7.4 INCREASED SILICATE-BASED PRODUCTIVITY

Despite a probable decrease in Si availability, the overall productivity of the ecosystem appears to have increased in response to the increased nitrogen load. This is evidenced by (1) equal or greater net silicate-based phytoplankton community uptake of silica in the mixing zone, compared to the 1950s (Turner and Rabalais 1994a), and (2) greater accumulation rates of biogenic silica (BSi) in sediments beneath the plume (Turner and Rabalais 1994b).

Bien et al. (1958) first documented the dilution and nonconservative uptake of silicate in the Mississippi River plume by sampling from the river mouth seaward in 1953 and 1955. A notable characteristic of the mixing diagram is that the concentration of silicate often falls below the conservative mixing line, thus indicating biological uptake. Uptake can be statistically modeled as a deviation from this mixing line, which Turner and Rabalais (1994a) did for 31 adequately sampled data sets obtained along a river to open Gulf salinity gradient. They found that the concentration of silicate at the 20 ppt mixing point declined in the last several decades during the winter–spring (January–April) and summer months (June–August); however, there was no discernible change during the fall–winter months (October–December). They normalized for the effects of varying concentrations in the riverine end-member (e.g., Loder and Reichard 1981) and compared the estimated net silicate uptake at 30 ppt as a function of silicate riverine end-member concentration (Figure 7.5). Nonconservative mixing (biological uptake) of silicate was indicated in all data collections. The net uptake (at 30 ppt) above dilution ranged from 1% to 19% of the intercept concentration, and the data groups for before and after 1979 were remarkably similar. Further, the net silicate uptake appears to be even higher after than before 1979 (Figure 7.5); these differences were apparent in the summer months during peak hypoxia development (June–August) as well. The results from this analysis suggested that net silicate uptake in the dilution gradient from river to sea has remained the same, or even increased, as the riverine concentration decreased.

Turner and Rabalais (1994b) documented that surficial sediments, directly downstream and beneath the surface riverine/estuarine dilution plume, reflected the *in situ* primary production and subsequent transport of organic carbon from surface to bottom waters within the Mississippi River Bight (Rabalais et al. 1992a; Turner and Rabalais 1994b) (Figure 7.3). They further quantified the silica in the remains of diatoms sequestered as biologically bound silica (BSi) in dated sediment cores from the same region. Relative changes in the % BSi reflect changes in *in situ* production (Conley et al. 1993). The pattern in % BSi in dated sediment cores paralleled the documented increases in nitrogen loading in the lower Mississippi River over the same period that silicate concentrations were decreasing (Turner and Rabalais 1994b). The increased % BSi in Mississippi River Bight sediments was direct evidence for the increase in flux of diatoms from surface to bottom waters beneath the Mississippi River plume.

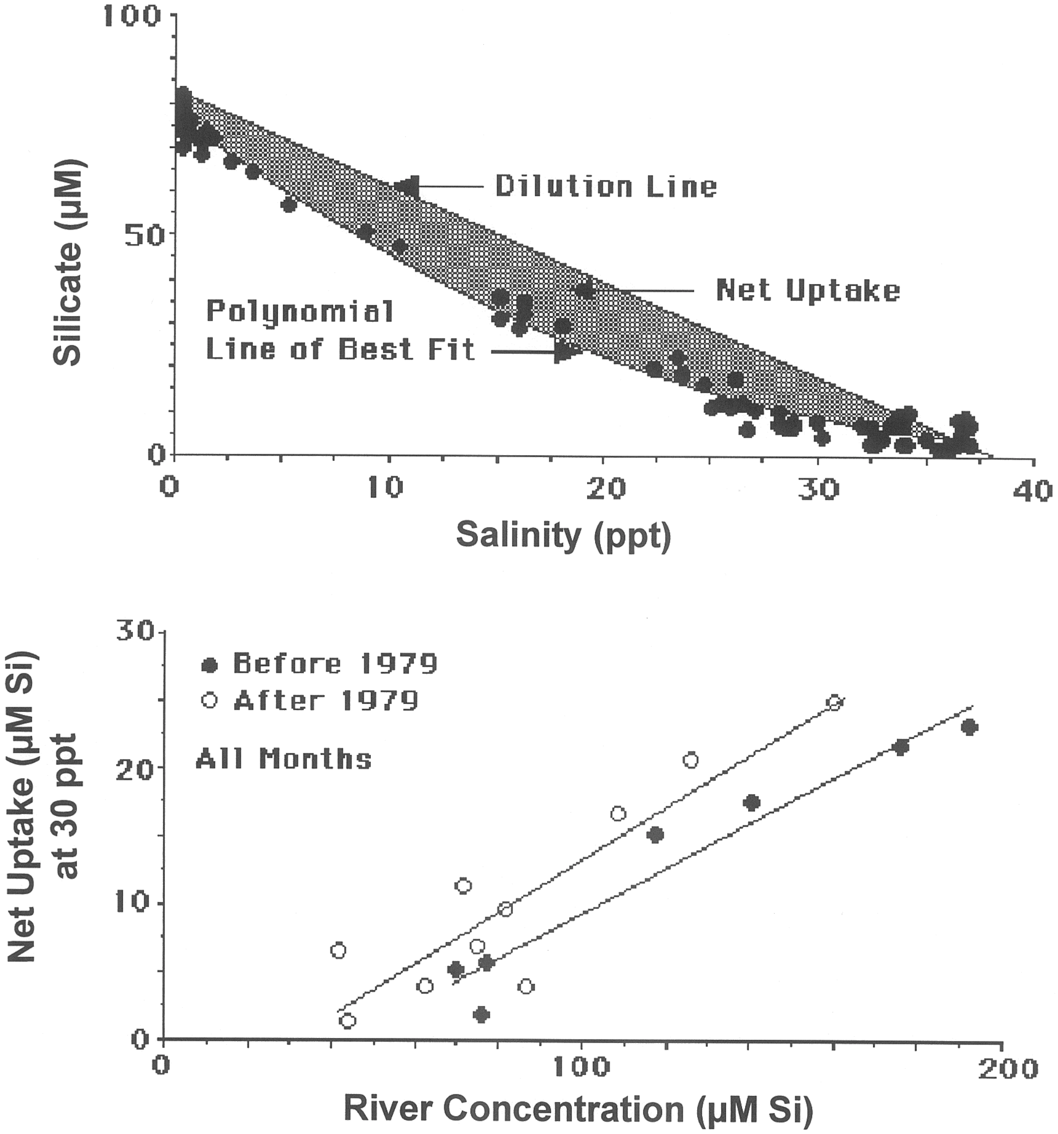


FIGURE 7.5. Upper panel: Example of the dilution of silicate in the Mississippi River plume (for June 1953) with sea water. Lower panel: Estimated net uptake of silicate above dilution losses at 30 ppt for all data but separated for time periods. (*Modified from Turner and Rabalais 1994a.*)

The highest concentrations of BSi were in sediments deposited in water 25–50 m deep in the middle of the sampling area. The % BSi in sediments from deeper waters (110 and 200 m) was generally stable through time, but rose in the shallower stations (10 and 20 m) around the beginning of this century. At intermediate depths (27–50 m), where both the % BSi concentration and accumulation rates were highest, parallel changes in the % BSi with time were evident in the two cores, especially in the 1955–65 period (a rise and fall) and a post-1975 or 1980 rise that was sustained to the sampling date (1989) (Turner and Rabalais 1994b) (Figure 7.6). The general pattern that emerged was a steady accumulation of BSi from 1800 to 1900, then a slow rise, followed by a more dramatic rise in the 1970s and 1980s. Diagenesis of the BSi undoubtedly occurred in these cores, but will be relatively low because the sedimentation rate was high (> 0.5 cm/yr). Furthermore, others have found the record of BSi to be a good indicator of *in situ* production. Conley et al. (1993) summarized for freshwater lakes that, in general, accumulation of BSi in sediments mimics overlying water column productivity, and that, the more diatoms that are produced by nutrient-enhanced growth, the more BSi will be deposited.

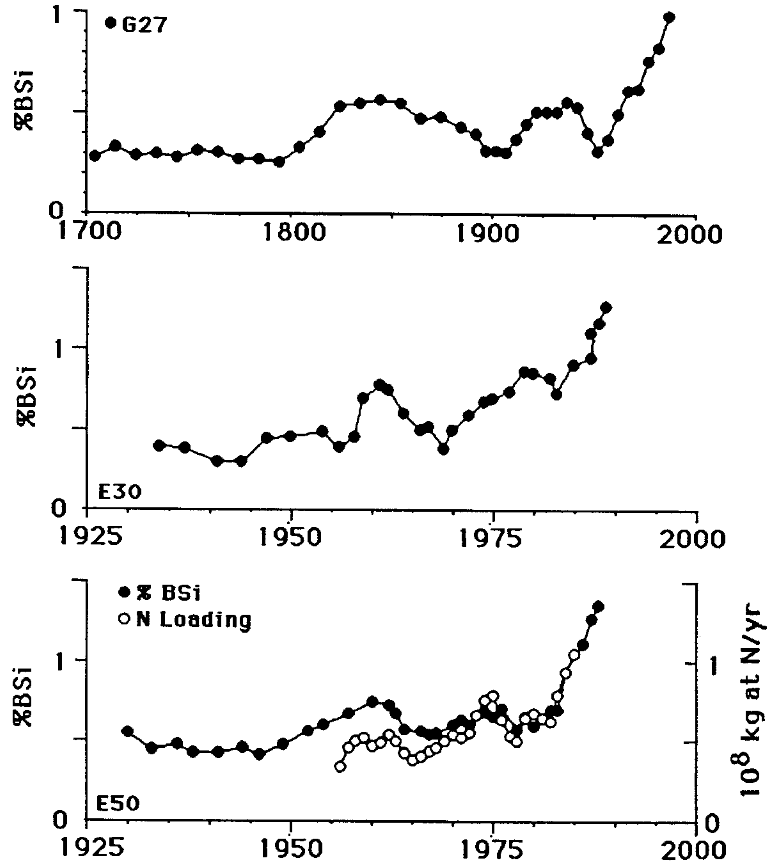


FIGURE 7.6. Average concentration of biologically bound silica (BSi) in sediments in each section of three dated sediment cores from stations in the Mississippi River Bight in depths of 27–50 m. NOTE: Stations are in Figure 7.4. A 3-year running average is plotted by time determined from Pb-210 dating. The figure for station E50 is superimposed with a 3-year average nitrogen loading from the Mississippi River. (*Modified from Turner and Rabalais 1994b.*)

Conley et al. (1993) predicted that coastal marine systems would not see long-term depletion of dissolved silicate with eutrophication, as in enclosed lakes, because regeneration from the sediments would maintain sufficient dissolved silicate levels to prevent silica limitation. The Turner and Rabalais (1994b) results that show continued accumulation of biologically bound Si in sediments beneath the plume and similar or higher net silicate uptake by phytoplankton in the plume surface waters, despite lower concentrations of Si delivered by the Mississippi River, support this hypothesis. Further changes in Si:N ratios, however, may result in more severe Si limitation than at present, and Si limitation may not be alleviated by benthic Si regeneration, especially during periods of strong stratification.

The increase in BSi in sediments from the mid-1850s to the early 1900s also supports the hypothesis of Mayer et al. (1998) that organic nitrogen associated with the suspended sediment load may have been a relatively large proportion of the total nitrogen load in river-dominated coastal regions prior to human activities that lead to suspended sediment reductions. Mobilization of sediments and associated organic nitrogen during clearing of the land during westward expansion may have resulted in enhanced productivity on the continental shelf. This productivity (if that is the case) stabilized and then increased since the 1950s, consistent with the increased load of dissolved inorganic nitrogen.

7.5 INCREASED PRODUCTIVITY

The organic accumulation in sediments in the middle of the Mississippi River Bight during the 1980s was 90 g C/m^2/yr, based on sedimentation rates and percent carbon of the sediments (Turner and Rabalais 1994b). This was approximately 30% of the estimated annual phytoplankton production (Sklar and Turner 1981; Lohrenz et al. 1990). If the assumption is made that the BSi:C ratio at the time of deposition remained constant this century, then the increased BSi deposition represented a significant change in carbon deposition rates (up to 43% higher in sections of cores dated after 1980 than those dated 1900–1960).

These results were corroborated by the same rate of increase in marine-origin carbon in sediment cores also collected within the Mississippi River Bight at site 10 (Eadie et al. 1994) near station E30 of Turner and Rabalais 1994b (Figure 7.4). Eadie et al. (1994) estimated accumulation rates of about 30 g C/m^2/yr in the 1950s to 50–70 g C/m^2/yr at present. The rate of burial was significantly higher at site 10 within the area of chronic hypoxia (approximately 70 g C/m^2/yr), in comparison to site 1, at which hypoxia was not documented (approximately 50 g C/m^2/yr). The δ^{13}C partitioning of organic carbon into terrestrial and marine fractions further indicated that the increase in accumulation of carbon in both cores was in the marine fraction (Figure 7.7). The accumulation of carbon in the sediments at station 10 of Eadie et al. (1994) was correlated with Mississippi River nitrate flux (Figure 7.8).

Sediment cores analyzed for different constituents (Turner and Rabalais 1994b; Eadie et al. 1994) documented eutrophication and increased organic sedimentation in bottom waters, with the changes more apparent in areas of chronic hypoxia and coincident with the increasing nitrogen loads from the Mississippi River system. The depletion of bottom-water oxygen—its persistence and areal coverage on the shelf—is thus indicated to have been altered this century as well.

7.6 CHANGES IN INDICATORS OF OXYGEN STRESS

Glauconite, a sediment mineral indicative of reducing environments and geologic anoxic settings, was used to document changes in hypoxia. The average glauconite abundance accounted for ~5.8% of the coarse fraction of sediments before a transition point in the early 1940s and accounted for ~13.4% after (Figure 7.9) (Nelsen et al. 1994). These data suggested that hypoxia existed at some level at site 10 before 1940 and that subsequent anthropogenic influences have exacerbated the problem. Conditions before 1900 cannot be determined from this core.

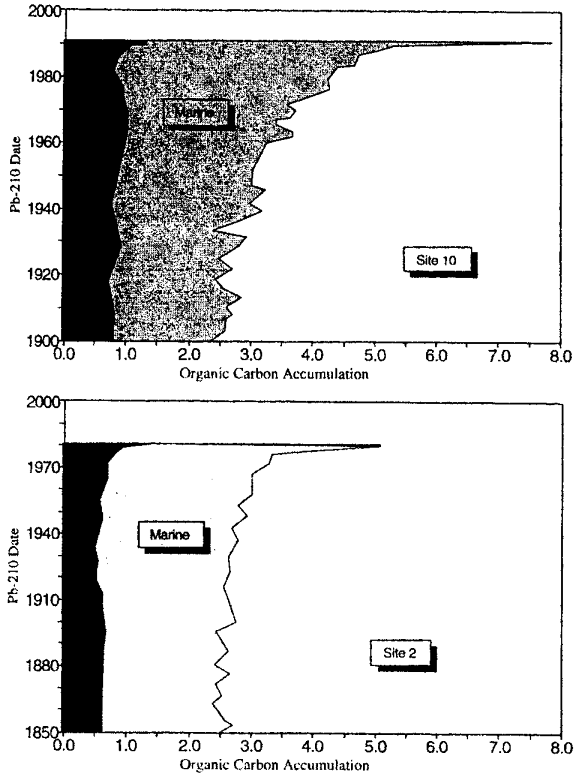


FIGURE 7.7. Organic carbon accumulation rates (mg C cm^2/yr) at stations 10 (near E30, Figure 7.4) and 2 (near D80 LaSER). NOTE: The partitioning of the organic matter into terrestrial (black) and marine (gray) fractions is based on $\delta^{13}C$. The recent increase in organic accumulation is much larger in the core within the region of hypoxia. Accumulation of terrestrial organic matter is also larger at station 10, implying it receives more river influence than station 2. (*From Eadie et al. 1994.*)

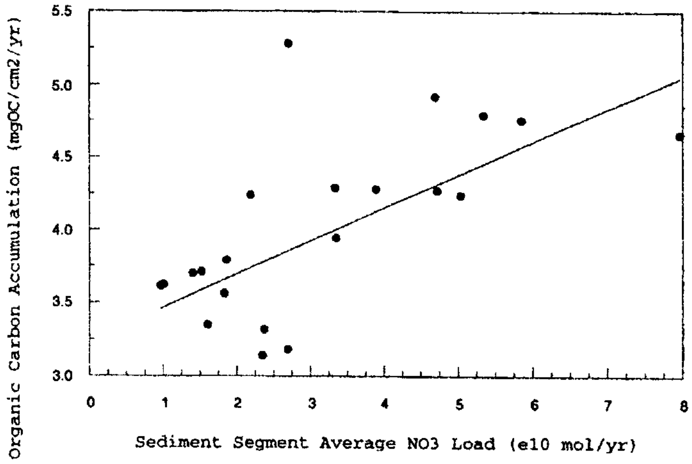


FIGURE 7.8. Correlation of the organic carbon accumulated in each sediment section versus the Mississippi River nitrate flux for the same (Pb-210) interval. NOTE: The least-squares fit for core 10 is y = 3.3 + 0.22 x (r^2 = 0.65). (*From Eadie et al. 1994.*)

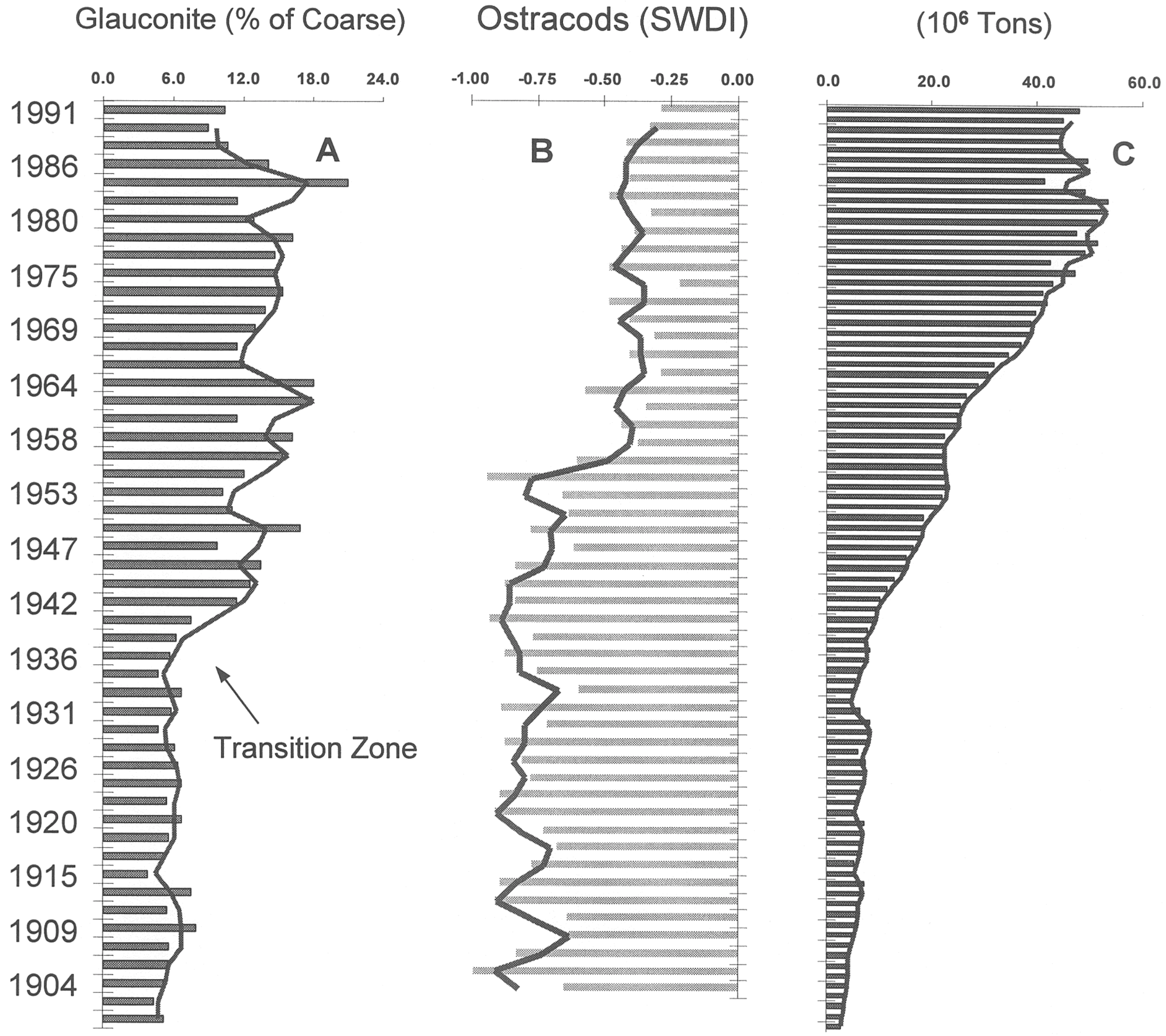


FIGURE 7.9. Glauconite grain abundance (A) and the Shannon–Wiener Diversity Index of ostracods (B) for station 10 compared to the application of fertilizer in the United States (C) for the years ~1900–1990. NOTE: A similar decline in SWDI for benthic foraminiferans was reported for selected sections of core 10 in Nelsen et al. (1994). (*Modified from Nelsen et al. 1994; Nelsen et al. unpublished data*.)

Dominance trends of benthic foraminifera and ostracods (single-celled organisms and metazoans, respectively, that produce a calcium carbonate shell that remains intact in buried sediments) were used as indicators of reduced oxygen levels and/or carbon-enriched sediments (Nelsen et al. 1994; Blackwelder et al. 1996; Sen Gupta et al. 1996; Rabalais et al. 1996; Nelsen et al. unpublished data). Sen Gupta et al. (1996) and Rabalais et al. (1996) examined several cores in the Mississippi River Bight (Figure 7.5). Nelsen et al. (1994) and Blackwelder et al. (1996) analyzed a core from site 10 near station E30 (Figure 7.4).

Trends of benthic foraminifera are useful as indicators of reduced oxygen levels and/or carbon-enriched sediments (Sen Gupta et al. 1981; Sen Gupta and Machain–Castillo 1993). The same series of Pb-210 dated sediment cores used for BSi analyses (Turner and Rabalais 1994b; Figure 7.4) were used to determine benthic foraminifera (Sen Gupta et al. 1996; Rabalais et al. 1996). Benthic foraminiferal density and diversity are generally low in this environment, but a comparison of assemblages in surficial sediments from areas differentially affected by oxygen depletion indicated that the dominance of *Ammonia parkinsoniana* over *Elphidium excavatum*. (A–E index) was much more pronounced under hypoxia than in well-oxygenated waters (Sen Gupta et al. 1996). The abundance of *A. parkinsoniana* was correlated with phytoplankton biomass in surface waters and % BSi (i.e., a food source indicator) in sediments. The A–E index also correlated strongly with the percentage of total organic carbon in surficial sediments. Thus, the index was affected by seasonal hypoxia produced by phytoplankton blooms that are recorded in the sediments in BSi and carbon content.

In the context of modern hypoxia, species distribution in dated sediment cores revealed stratigraphic trends in the *Ammonia/Elphidium* ratio that indicated an overall increase in oxygen stress (in intensity or duration) in the last 100 years (Figure 7.10). In particular, the stress seemed especially severe since the 1950s. It was notable that there was no trend in the A–E index for station G50 outside the zone of persistent hypoxia and that the index in 1988 for station C10 fell off the trend line (i.e., no low oxygen during the mid-summer 1998 cruise).

In the last 100 years, both *A. parkinsoniana* and *E. excavatum* became less important components of the assemblage, while *Buliminella morgani* showed an unusual dominance (Figure 7.10 and Blackwelder et al. 1996). *B. morgani*, a hypoxia-tolerant species, is known only from the Gulf of Mexico and dominates the population (> 50%) within the area of chronic seasonal hypoxia (Blackwelder et al. 1996). It increased markedly since 1900 in the sediments analyzed by Blackwelder et al. (1996) and since 1800 for station G27 of the Sen Gupta et al. (1996) study (Figure 7.10). *Quinqueloculina* sp. (a significant component of the modern assemblage only in well-oxygenated waters) has been absent from the record of the G27 core since the 1870s, but was a conspicuous element of the fauna in the previous 200 years. The historical absence of *Quinqueloculina* sp. since 1900 at station G27 corroborates the presence of glauconite at site 10 since 1900 (Nelsen et al. 1994). The occurrence of *Quinqueloculina* sp. at station G27 before 1900, however, indicates that oxygen stress was not a problem before then. A decided decrease in the Shannon–Wiener diversity index since the early 1950s for foraminiferans (not illustrated) and ostracods (Figure 7.9) was documented for site 10 near E30 in the present-day hypoxic zone.

These several surrogates for actual oxygen measurements indicate an overall increase in oxygen stress (in intensity or duration) in the last 100 years. Oxygen stress appears to be especially severe since the 1950s, is coincident with the onset of increases in riverine nitrogen loading, and parallels the increased carbon flux to the seabed.

7.7 LINKAGES OF LONG-TERM COASTAL ECOSYSTEM CHANGES WITH MISSISSIPPI RIVER CHANGES

Mississippi River nutrient concentrations and loadings to the adjacent continental shelf changed dramatically this century, with an acceleration of these changes since the 1950s–60s. Concentrations of dissolved N and P doubled and Si decreased by 50%; the dissolved Si:N ratio dropped from 4:1 to 1:1; and nitrate concentration peaks in the spring, where it did not historically. The resulting nutrient composition in the receiving Gulf waters shifted toward stoichiometric nutrient ratios closer to the Redfield ratio. N and P now appear to be less limiting for phytoplankton growth, while some increase in Si limitation is probable. Despite a decrease in Si availability, the overall productivity of the ecosystem in the Mississippi Bight (in both diatoms and total phytoplankton) increased coincidentally with increased nitrogen loads. Finally, an analysis of benthic foraminifera and ostracods indicates an increase in oxygen deficiency stress within the Mississippi Bight this century, with a dramatic increase since the 1950s.

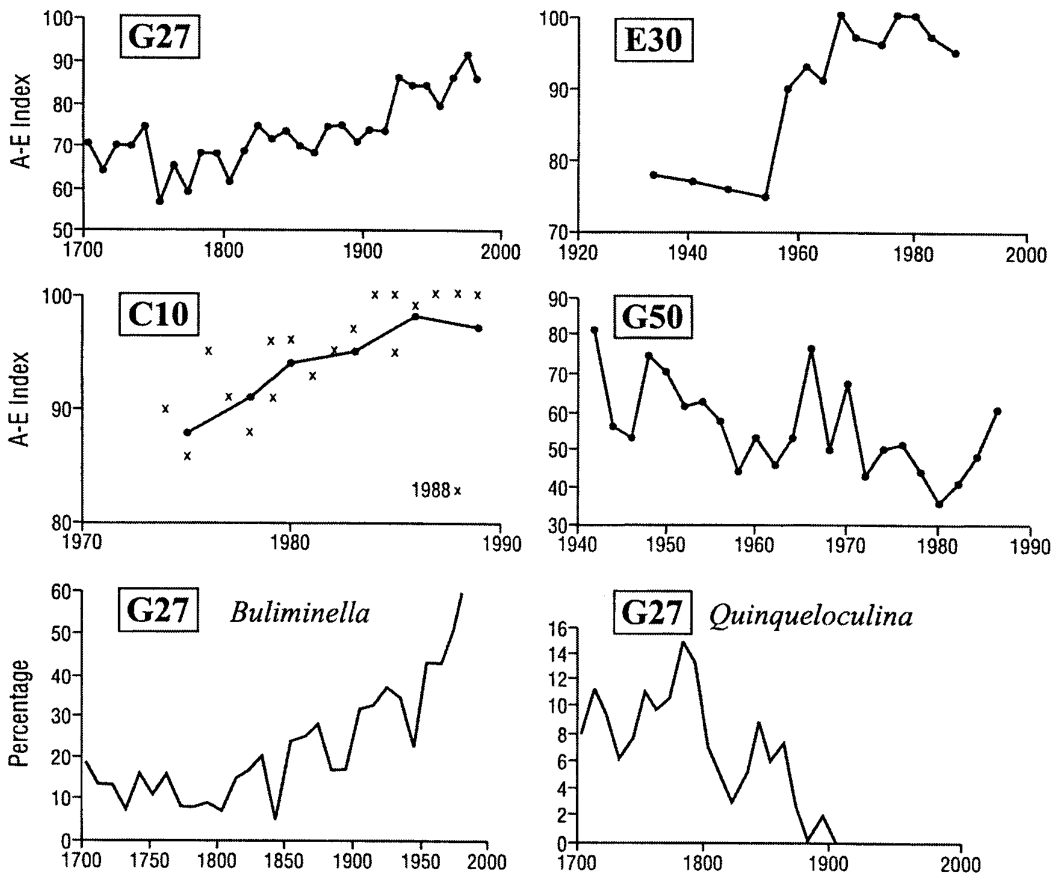


FIGURE 7.10. Changes in benthic foraminiferan species with stratigraphic depth in Pb-210 dated sediment cores from stations in the Mississippi River Bight. NOTE: A line connecting three-year averages is superimposed on the data for C10. (*Modified from Rabalais et al. 1996; Sen Gupta et al. 1996.*)

The importance of the water column's physical structure to the development and persistence of hypoxia is clear, but the discharge of the Mississippi River (i.e., amount of flow) through the birdfoot delta since the 1800s has been relatively constant aside from normal decadal scale variations. Increasing capture of Mississippi River water by the Atchafalaya River, until stabilized at 30% in 1977 by the U.S. Army Corps of Engineers, most likely increased the strength of stratification down-plume (to the west) of the Atchafalaya effluent and may have aggravated hypoxic conditions there. Dated sediment cores do not exist for that area, nor do sediments accumulate such that similar studies can be conducted there.

A corollary to a scenario of worsening hypoxia down-plume from the Atchafalaya might be the potential for less hypoxia in the Mississippi River Delta Bight as more water went to the Gulf via the Atchafalaya. Delivery of discharge through the birdfoot delta, however, has been altered this century as well. More flow has been sent down Southwest Pass—even with the Atchafalaya receiving more, as opposed to Main Pass, South Pass, and Pass a'Loutre—to maintain navigational access. Thus, the Mississippi River Delta Bight has continued to be influenced by a significant proportion of the discharge through the birdfoot delta. The slight increasing trend for the main channel discharge (Bratkovich et al. 1994) occurred during September through December—a period less important to the formation and maintenance of hypoxia.

The Mississippi River Bight provides the best available information for long-term changes on the Louisiana continental shelf. Although flows (measured at Vicksburg, Mississippi) have been stable for decades, the long-term delivery among distributaries and passes has been unstable. Despite the relative change in discharge through the many distributaries, most of the water flows west along the Louisiana coast and influences the area where hypoxia develops. Given the importance of stratification to the development and maintenance of hypoxia and the relationship of nutrient flux to river discharge, it is tempting to look at short-term changes in discharge volume and size of bottom-water hypoxia in mid-summer to identify relationships or trends in hypoxia. As pointed out in Chapter 3, the area estimates from mid-summer cruises should not be overinterpreted in making annual comparisons or identifying trends, and the relationship of the estimated mid-summer area and mean Mississippi River discharge is not robust.

The best information from which to draw empirical relationships to hypoxia is derived from the more frequently collected data on transect C and at station C6*, and the long-term data provided by dated sediment cores from the Mississippi River Delta Bight. The transect studies demonstrate time and spatial lags of net production, carbon flux, and development of hypoxia in relation to freshwater discharge and nutrient flux. There is much variability, but the long-term seasonal pattern is evident.

To determine how the marine ecosystem has changed over longer periods, we must rely on surrogates for surface-water production and hypoxia in sediments. There is no doubt that the concentration and flux of nitrogen have increased since the 1950s–60s, particularly in the spring, and that the flux of nitrogen per unit of volume discharge is greater now than historically. The patterns of biogenic silica (a surrogate for diatom production) and carbon accumulation in the Mississippi River Bight clearly follow nitrogen loading, which has increased disproportionately in relation to annual variability in river discharge. On the other hand, the long-term pattern of freshwater discharge is comparatively stable. Thus, the observed changes in biological responses are probably not due to changes in the amount or distribution of freshwater runoff and resultant stratification, but rather to increased nitrogen loads.

The distributary landscape for delivery of water to the continental shelf, both through the Mississippi River watershed and at the deltas, has changed over several centuries. Construction of navigation and flood-control levees along the lower Mississippi River system has potentially restricted the amount of water that flows over banks and through riparian and coastal wetlands, where some removal of nutrients is likely. Turner (1998a), however, estimated that less than 10% of the historically low nutrient concentrations of the Mississippi River (prior to 1950) could be reduced before reaching the Gulf of Mexico by overland flow within the lower reaches of the river and the deltaic wetlands. Turner also calculated that the reduction of overbank flooding from flood protection did not significantly diminish nutrient loading to the continental shelf. Leveeing of the lower Mississippi River corridor was completed in the early 1920s, and most of the changes in the continental shelf occurred since the 1950s, when nutrient loads and concentrations changed dramatically.

It is likely that multiple landscape changes throughout the Mississippi River system from the early to mid-1800s have contributed to changes in the ecology of the adjacent continental shelf ecosystem. These landscape changes, while they likely contribute to worsening eutrophication and oxygen stress on the Louisiana shelf, are diffuse, cumulative, and difficult to document. The most dramatic changes in eutrophication and oxygen stress have occurred since the 1950s, when nutrient delivery to the Gulf of Mexico changed two- to threefold over previous decades.

Understanding the relationships of hypoxia to the physical setting, biological processes on the continental shelf, and nutrient delivery by the Mississippi River is important both in eventual and relevant management of nutrients and in implementation of landscape changes that may minimize the extent, duration, or severity of hypoxia on the adjacent continental shelf. Similarly, it is important to identify those factors that influence the development and maintenance of hypoxia or a worsening of the conditions that are attributable to human activities in the watershed. Some are controllable; others are not.

CHAPTER 8

Worldwide Perspective

Many coastal ecosystems around the world suffer from eutrophication and hypoxia (see the Topic 2 report by Diaz and Solow 1999; Diaz and Rosenberg 1995). Diaz and Rosenberg (1995) documented that many systems are hypoxic now that were not historically, and others have expanded the geographic extent or have increased in severity—in lower dissolved oxygen concentrations, in prolonged periods of exposure, or both. Long-term increases in nutrient concentrations in coastal waters, along with increased primary production, have been documented elsewhere in the world, e.g., the Baltic Sea (Larsson et al. 1985; Rosenberg 1986; Wulff and Rahm 1988); the Kattegat and Skaggerak (Rosenberg 1986; Andersson and Rydberg 1987); the sounds separating Sweden from Denmark (Rosenberg 1986); the northwestern shelf of the Black Sea (Tolmazin 1985); the northern Adriatic Sea (Faganeli et al. 1985; Justić et al. 1987); and the Dutch coast of the North Sea (Fransz and Verhagen 1985). In their opinion (Diaz and Rosenberg 1995), no other environmental stressor has changed to the degree that oxygen depletion has changed in the last several decades.

Smaller and less frequent zones of hypoxia than that of the northern Gulf of Mexico occur in U.S. coastal and estuarine areas, e.g., the New York Bight (Garside and Malone 1978; Swanson and Sindermann 1979; Falkowski et al. 1980; Swanson and Parker 1988); Chesapeake Bay (Officer et al. 1984; Malone 1991, 1992; Boynton et al. 1995); Long Island Sound (Welsh and Eller 1991; Welsh et al. 1994; Parker and O'Reilly 1991); Mobile Bay (Loesch 1960; May 1973; Turner et al. 1987); and the Neuse River estuary (Paerl et al. 1998). Where sufficient long-term data exist (e.g., Chesapeake Bay) there is clear evidence for increases in nutrient flux, increased primary production, and worsening hypoxia.

Thorough analyses of multiple indicators in sediment cores from Chesapeake Bay indicate that sedimentation rates and eutrophication of bay waters have increased dramatically since the European settlement of the watershed (Cooper and Brush 1991, 1993; Cooper 1995). Results also indicate that hypoxia and anoxia may have been more severe and of longer duration in the last 50 years than in the bay's previous history. Changes were documented in biogenic silica, pollen, diatom remains, estimates of degree of pyritization of iron, and increases in chlorophyll and N:P ratios. The sediment core findings corroborate long-term changes in Chesapeake Bay water column chlorophyll biomass since the 1950s (Harding and Perry 1997). The parallels of the Chesapeake Bay eutrophication and hypoxia to those of the Mississippi River watershed and Gulf of Mexico hypoxia are striking, in particular those of the last half century. Landscape alteration began much earlier in Chesapeake Bay than in the Mississippi River watershed.

Comparative analyses of riverine nutrient concentrations and loads and ecosystem-level changes in adjacent coastal waters were documented by Justićet al. (1994, 1995b) for the Po River/northern Adriatic Sea and the Mississippi River/northern Gulf of Mexico. Nitrogen and phosphorus loads to both rivers have increased significantly during the last 30 years, and their relative proportions and proportions to silicate have also changed. Nutrients in both riverine discharges are more balanced and overall less limiting for phytoplankton growth. There is evidence from both systems that primary productivity in surface waters has increased under these conditions and that bottom-water oxygen concentrations have decreased. Numerical comparisons of the two systems can be seen in comparisons of oxygen budgets (Figure 8.1). The evidence is clear for the northern Adriatic that the frequency of oxygen surplus at the surface > 1 ml/l has significantly increased from 6% in 1955–66 to 13% in 1972–82. The frequency of occurrence of bottom oxygen deficit > 1 ml/l has also changed significantly from 19% to 33% over the same interval. Similar historic data are not available for the northern Gulf of Mexico hypoxic zone, but the sedimentary record clearly indicates an increase in oxygen stress. For comparative purposes, the changes in oxygen surplus and deficit in the northern Adriatic are compared to the current relative frequency in the northern Gulf of Mexico for 1985–91 (Figure 8.1). The Gulf data show a surface oxygen surplus of > 1 ml/l in about 21% of the cases, while the bottom oxygen deficit of > 1 ml/l is about 87% of the cases, compared to similar frequencies of 13% and 36% for the northern Adriatic.

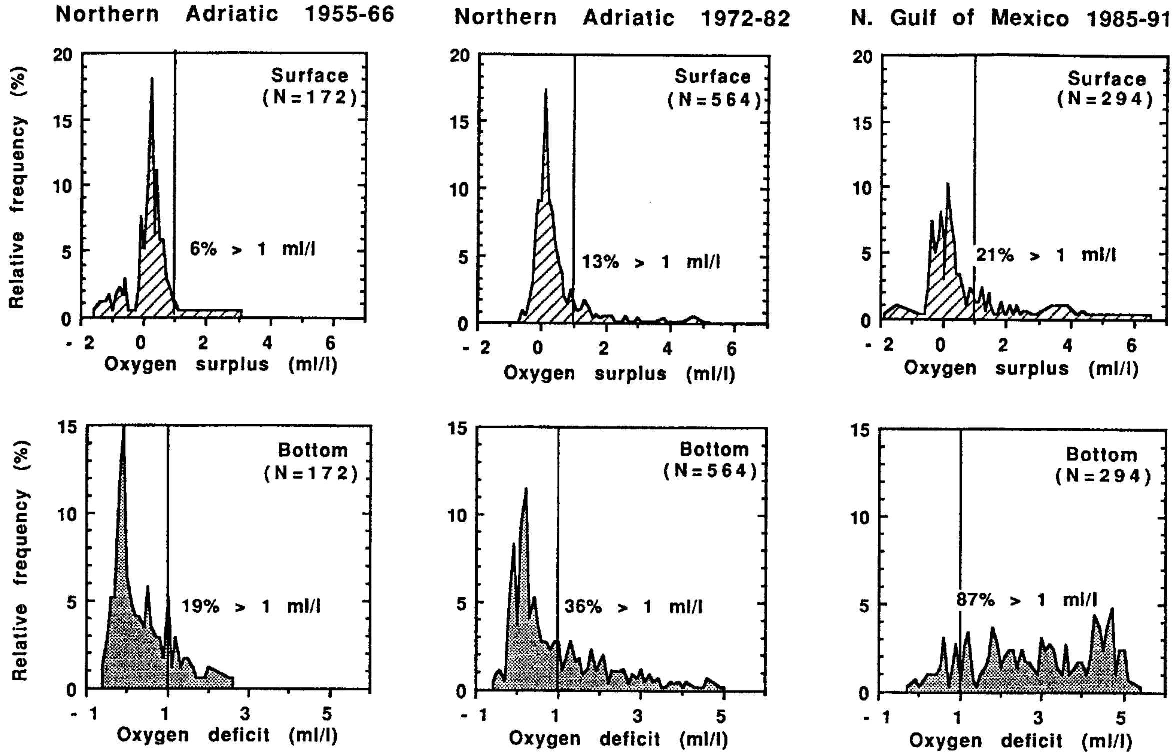


FIGURE 8.1. Relative-frequency spectra for surface oxygen surplus and bottom oxygen deficit in the northern Adriatic Sea (1955–65 and 1972–82) and in the northern Gulf of Mexico (1985–91). NOTE: Vertical lines indicate an arbitrary value of +1 ml/l. Station locations and methods are in Justić et al. (1994). (*From Justićet al. 1994.*)

An analysis of seasonal oxygen cycling in the northern Adriatic between 1911 and 1982 (Justić et al. 1994) (Figure 8.2) shows that both the relative surface oxygen surplus and the relative bottom oxygen deficit have increased through time, indicating that the conditions are moving from oligotrophy toward eutrophy (Justić 1991b). The recent data tend to scatter below the line indicating a 1:1 relationship, which means that the ecosystem in the northern Adriatic is in transition toward heterotrophy. Recurring summer hypoxia is more severe in the northern Gulf of Mexico, which is predominantly heterotrophic throughout the year and clearly more eutrophic than the northern Adriatic (Figure 8.2).

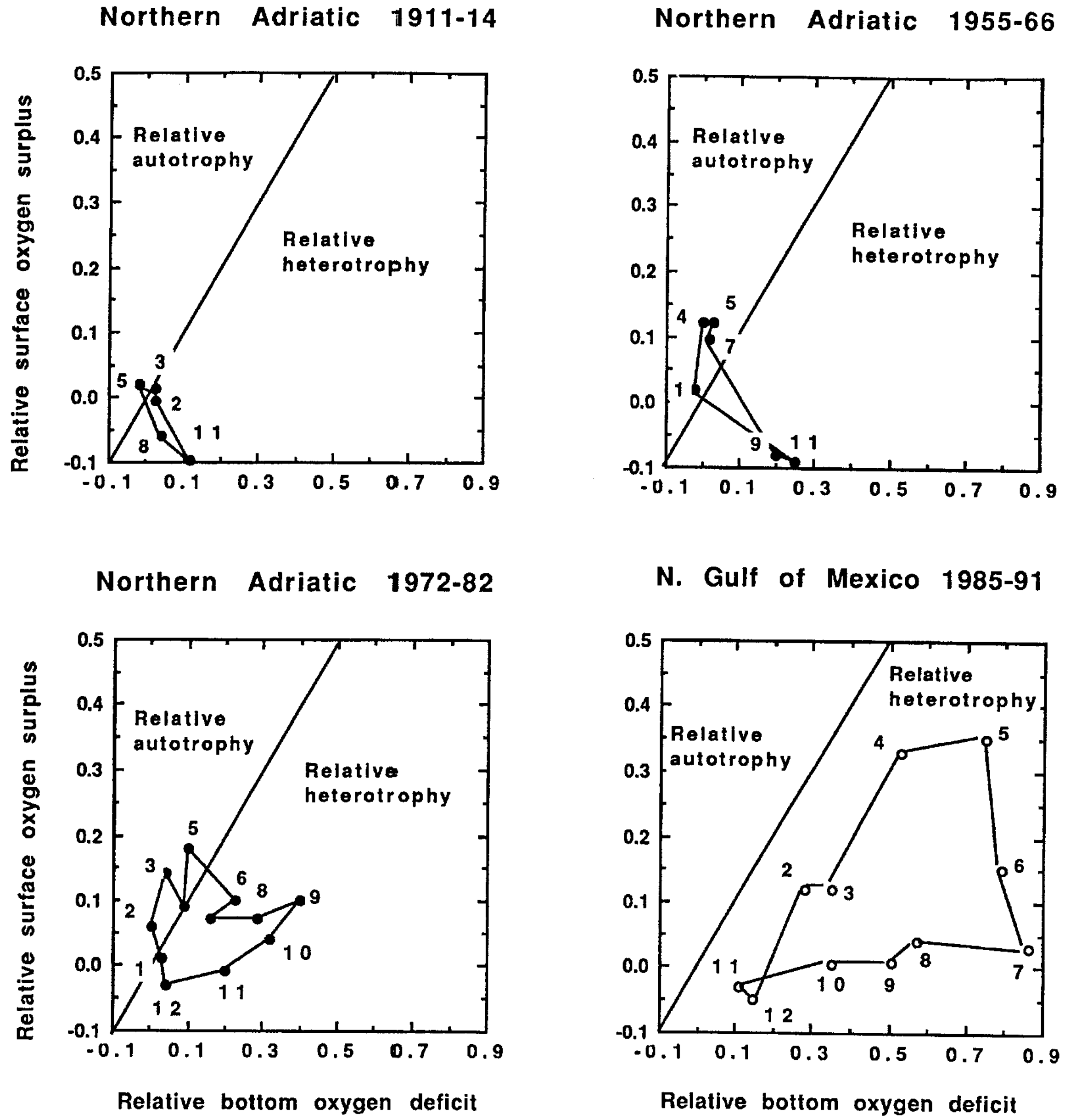


FIGURE 8.2. Changes in relative surface oxygen surplus (x_s) and relative bottom oxygen deficit (x_b) in the northern Adriatic Sea and in the northern Gulf of Mexico during an annual cycle. Note: Symbols are monthly averages for the locations and time periods indicated. A 1:1 relationship is shown and separates relative autotrophy and relative heterotrophy along a gradient of oligotrophy to eutrophy. Station locations and methods are in Justić et al. 1994. (*From Justićet al. 1994.*)

CHAPTER 9

Scenarios of Change—The Future

9.1 CLIMATE CHANGE

The northern Gulf of Mexico is a coastal area likely to experience increased freshwater runoff as a result of global climate change. Miller and Russell (1992) used a general circulation model to predict increased precipitation of about 20% in the Mississippi River watershed, with runoff expected to increase in most months, particularly from May through August. The resulting higher runoff in summer would likely affect water column stability, surface productivity, and oxygen cycling in the northern Gulf of Mexico, similar to the flood of 1993.

Justić et al. (1996) superimposed a 20% increase in freshwater runoff during May–August to estimate monthly average runoff from the Mississippi River for the period 1985–92. Manipulations of their physical-biological two-box model (Justić et al. 1996), with depressed surface-water salinity and increased net production, revealed that there would be a 30–60% decrease in summertime subpynoclinal oxygen content at reference station C6* relative to the 1985–92 average. The resulting seasonal decline in bottom-water oxygen levels in the model (Figure 9.1) very closely paralleled the actual sequence of events in 1993, when freshwater inputs and nutrient flux, especially nitrate and silicate, remained at elevated levels following the spring peak, as opposed to declining to the usual summertime lows.

The same model was used to examine increases in nutrient flux from the Mississippi River system in relation to net production of the system and bottom-water oxygen deficiency (Justić et al. 1997). Model simulations for 1993 compared to the 1985–92 calibration data indicated that the surface-water net production at station C6* increased by 26%. Given similar carbon flux estimates and oxygen uptake rates in the lower water column for 1993 compared to 1985–92, less of the organic production was decomposed in the lower water column and sediments in 1993. Thus, more carbon (65 g C/m^2/yr relative to the 1985–93 average, see Figure 6.17 in Section 6.9) was available for burial and accumulation in the sediments. This projected increase is of the same magnitude as the one that has occurred since the 1940s due to the introduction of anthropogenic nutrients (Turner and Rabalais 1994b; Eadie et al. 1994).

An increase in annual net production of 32 g C/m^2/yr was observed during the flood of 1993, thus indicating the general validity of a doubled CO_2 scenario consistent with long-term changes in coastal water productivity (Turner and Rabalais 1994b) and measured or calculated carbon accumulation rates (Turner and Rabalais 1994b; Eadie et al. 1994). The total oxygen uptake in the lower water column (10–20 m), in contrast, is likely to remain the same at its present value of about 200 g O_2/m^2/yr, because the present total oxygen uptake in the inner section of the hypoxic zone has already reached the limit that is set by the availability of dissolved oxygen. Thus, carbon export and burial, rather than *in situ* respiration, are likely to be the dominant processes balancing coastal carbon budgets, leading perhaps to an expanded extent of the hypoxic zone, as seen in the model prediction (Figure 9.1) and under flood conditions in 1993.

Climate change will most likely affect other large-scale meteorological conditions, such as frequency and severity of tropical storms. Additionally, increased rainfall or temperature could affect nitrogen flux through cropping changes or in-stream processing of nutrients. Further analyses of offshore ecosystem responses and continued modeling efforts are warranted, with a refined assessment of climate change.

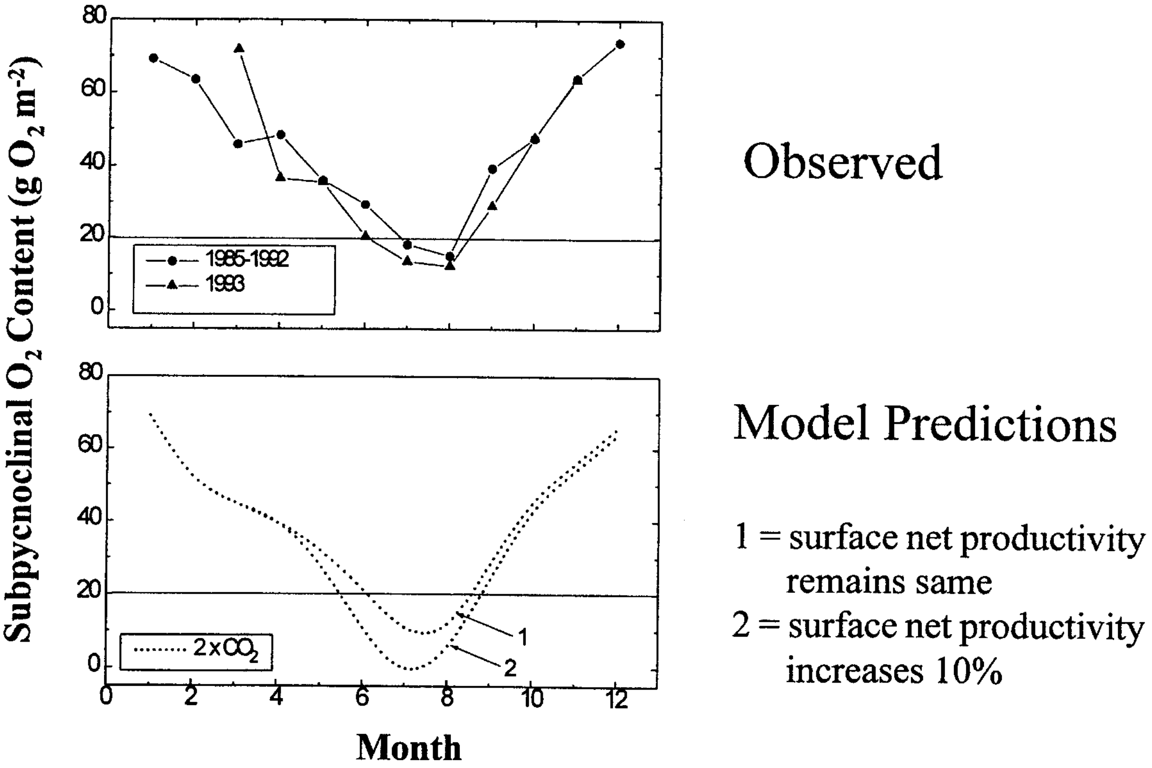


FIGURE 9.1. Seasonal changes in the oxygen content of the lower water column during 1985–92 and 1993 (top panel), and model predictions for a doubled CO_2 climate (bottom panel). NOTE: Two model predictions for total oxygen content of the lower water column are shown: (1) integrated net production remains the same as during 1985–92; and (2) the net production value for the May–August period is increased by 35%. (*From Justićet al. 1996.*)

9.2 PREDICTIONS

Changes in riverine end-member nutrient concentrations and ratios may affect coastal ecosystems in several ways. If nitrogen is in excess supply, diatoms often out-compete other algae if adequate silicate is available. When nitrogen increases and silicate decreases, flagellates may increase in abundance (Officer and Ryther 1980) and form blooms. Noxious blooms of flagellates, sometimes toxic, are increasingly common in coastal systems (Hallegraeff 1993). Zooplankton, the main diatom predator and a staple of juvenile fish diets, are thus affected by these nutrient changes in a cascading series of interactions (Turner et al. 1998). These interactions may also contribute to the formation of hypoxic water, which is unsatisfactory fish habitat (Coutant and Benson 1990).

An important consequence of eutrophication may be when a bloom occurs. Townsend and Cammen (1988) point out that the timing of the spring phytoplankton bloom in high latitudes affects significant benthic–pelagic couplings important to fisheries recruitment. Although the timing of a bloom is primarily determined by light conditions, an adequate nutrient supply is also required. Changes in the amount and timing of nutrient loading could affect recruitment success through a mismatch of larval recruitment and food supply, as well as an altered food chain. Early blooms with a greater sedimentation to the benthos could positively affect demersal fishes, but late blooms positively affect pelagic fisheries through a zooplankton food chain.

Rabalais et al. (1996) predicted how nutrient concentrations and ratios might change under a variety of management and economic schemes, with subsequent consequences to the marine environment. Their predictions assumed that stabilization would be reached in the Mississippi River Basin between the sequestering of diatoms from phosphorus-enhanced stimulation of blooms and regeneration of silicate and that silicate concentration in the river discharge would slowly increase over time. At present, and since the late 1980s, the Si:N ratio fluctuates near 1:1 (the Redfield ratio), and silicate and nitrate concentrations fluctuate near 100 μM. It is not clear whether the silicate concentration will increase as predicted.

Nutrient management scenarios may alter the amount of nitrogen and phosphorus entering the Gulf of Mexico. In a scenario of no change in current practices, nitrogen loading may plateau at the current 130 x 10^6 kg atoms N/yr. This would most likely perpetuate the current severity of hypoxia on the Louisiana shelf. An increase in nitrogen resulting from increased anthropogenic inputs (e.g., increased cropland in cultivation, fertilizer applications, atmospheric deposition, sewage inputs, animal wastes) would clearly aggravate the hypoxia problem. A reduction in nitrogen loading, particularly during the period of peak spring discharge, will return the system to a historic status of less production, carbon flux and accumulation, and hypoxia, but the length of time for the system response is unknown.

Managing for a single nutrient is difficult for the large Mississippi River system because N and P may change together but not linearly, and Si is complexly interrelated with P dynamics and/or water retention in the watershed. Nitrogen is the limiting nutrient for overall productivity in the northern Gulf of Mexico (Table 6.6), but other nutrients at times can occasionally become limiting. Deficiency of Si in relation to N, and possibly P, can result in compensatory qualitative changes, including new phytoplankton communities with noxious or toxic species. The potential direct and indirect effects of decreasing or increasing nitrogen, while ratios of Si:N fluctuate above or below the Redfield ratio of 1:1, are listed in Table 9.1. These are hypotheses based on documented responses of the system to changes in nutrient loads and ratios over the last several decades (Rabalais et al. 1996; Turner et al. 1998). The scenarios in Table 9.1 are clear that reducing N loading will decrease hypoxia, along with other possible effects when the Si:N atomic ratio exceeds that of Redfield.

Two models link Mississippi River discharge and nutrient flux to Gulf of Mexico hypoxia (Bierman et al. 1994; Limno-Tech, Inc. 1995; Justić et al. 1996, 1997). One is a mass-balance model of primary production and oxygen depletion, and the other is a coupled biological–physical two-box model with a temporal component. Each was developed based upon the long-term data set of Rabalais et al. (1991, 1996, 1998), and each has limitations as to parameterization, calibration, and ability to predict spatial and temporal responses to changes in nutrients. Still, useful simulations and experiments have been conducted with each that show responses of the coastal ecosystem to changes in riverine discharge and nutrient loads that are verifiable with subsequently collected hydrographic data. What these models demonstrate is the worsening of oxygen depletion in bottom waters with increased freshwater discharge and nitrogen load, or conversely, a reduction in oxygen demand in the lower water column with a reduction in nitrogen load to the surface waters. In other words, hypoxia in the northern Gulf of Mexico can be alleviated measurably by a reduction in the nutrient load. The degree of nutrient limitation needed to affect substantial temporal and spatial diminishment in the hypoxic zone cannot yet be determined from the existing models. Alternatively, the potential for hypoxia to worsen with increasing nutrient loads is also a very likely scenario, whether due to increased loads from agriculture, the atmosphere, or point sources or from climate change (Table 9.1; Rabalais et al. 1996; Justić et al. 1996, 1997).

9.3 NUTRIENT INTERVENTION

There are several success stories for improvement of estuarine and coastal ecosystems in response to nutrient abatement in the watershed or in direct discharges to the system. Nutrient management and intervention to reduce nutrient loads, particularly phosphorus, in Tampa Bay have met with successes in ecosystem restoration, including improved water clarity, reduced instances and biomass of cyanobacterial blooms, expansion of submerged aquatic vegetation (seagrass beds), increased catch of seagrass-dependent fishes (such as the highly valued commercial and recreational speckled sea trout), and improvement in dissolved oxygen conditions in bottom waters (Johansson and Lewis 1992). Retention reservoirs and weirs in upstream channels of Bayou Texar near Pensacola, Florida, were coupled with improved sewage treatment for improved water quality. There was almost a total reduction in fish kills, a 90% reduction in phytoplankton primary production, a virtual elimination of algal blooms, and re-establishment of public use (Moshiri et al. 1981). Nutrient reductions by half have lead to a decrease in the frequency of red tides in Japan’s Seto Inland Sea (Cherfas 1990). Sewage improvements led to water quality improvements in Kaneohe Bay, Hawaii (Smith 1981). There are many examples of small-scale hypoxia reversals associated with improvements in treatment of sewage and pulp mill effluents (Rosenberg 1972, 1976).

The magnitude of restoration needed to affect changes in much larger coastal systems with much larger watersheds, such as the Chesapeake Bay, Long Island Sound, the Baltic Sea, and the northern Gulf of Mexico, is daunting. Still, multi-state and multi-national agreements and cooperation are aimed at just that in the case of the first three.

Chesapeake Bay has been the focus of both intensive research on cultural eutrophication, including hypoxia, and extensive efforts to reduce nutrient inputs responsible for it (Malone et al. 1993; Boesch et al. in press). Boesch et al. (in press) outlined the process of developing and implementing nutrient controls in Chesapeake Bay. Although the causes, trends, and effects of eutrophication were incompletely known at the time, a regional commitment was made in 1987 by the states in the Chesapeake Bay watershed to reduce inputs of nitrogen and phosphorus by 40% by the year 2000. Subsequent research and modeling justified this commitment, and the reduction by 40% of those nutrient sources deemed controllable would improve water quality, but less than originally thought.

Strong public support and political commitment have allowed for progress in reducing nutrient inputs to Chesapeake Bay, particularly from point sources. The information to date is encouraging in that reduced concentrations of N and/or P in stream flow have been observed in several major rivers. In the open waters of the bay, however, there have been no statistically significant trends in nutrient concentrations. On the other hand, nutrient concentrations have declined in

TABLE 9.1. Possible scenarios for primary production, food web dynamics, and hypoxia in relation to changes in nitrogen loading (higher and lower) and Si:N ratios < 1:1 and > 1:1.

Assessment Factors	**High N Loading (higher than present)**	**Low N Loading (circa 1950s)**
Si:N < 0.8:1		
Primary Production	High	Low, but higher than receiving waters
Diatoms	Low	Low
Bloom Strength	High	Low
Bloom Frequency	Episodic	Almost none
Bloom Quality	Many toxic and/or noxious	Almost no toxic blooms
Hypoxic Area	Severe at times, more sporadic; expand into Texas regularly	Contracted in size
Fisheries		
Bottom feeders	Catastrophic loss	Catastrophic loss Loss
Pelagic	Replacement and loss	and some replacement
Commercial Fisheries	Very disruptive; gear changes; more travel time; irregular stocks; some collapses; health warnings	Catastrophic loss? Species replacement
Recreational Fisheries	Smaller-sized; many species shifts; frequent health warnings	Very disruptive
Si:N > 1.5:1		
Primary Production	Very high	Low
Diatoms	Dominant	Dominant
Bloom Strength	High and sustained	Low
Bloom Frequency	Episodic	Almost none
Bloom Quality	Some toxic and/or noxious; dominated by *Pseudo-nitzschia* spp.	Almost no toxic blooms
Hypoxic Area	Widespread, severe, persistent; expand into Texas regularly	Contracted in size
Fisheries		Revival of 'K'
Bottom feeders	Catastrophic loss 'r'	selected species Revival of 'K'
Pelagic	selected species gain 'K'	selected species
	selected species loss	
Commercial Fisheries	Disruptive, gear changes, more travel time, irregular stocks; some collapses; health warnings	More stable than present (assuming same effort and technology)
Recreational Fisheries	Smaller-sized, some species shifts; health warnings	Large specimens possible; more stable than present

several tidal rivers where significant and demonstrable load reductions have been achieved. The monitoring data do not reveal any significant changes to date in dissolved oxygen concentrations in areas of summer hypoxia of deep bottom waters. In these areas (typically the middle reaches of the mainstem and several major tributaries), generally either modest or no significant improvements have been noted in nutrient concentrations or algal biomass. Because progress in load reductions for N have generally lagged behind those for P, and recent high freshwater loads have delivered higher nonpoint-source inputs of nutrients, it is reasonable to expect that several more years may be required to realize improvements in Chesapeake Bay's hypoxia problems. On the other hand, there are some encouraging signs of recovery in submersed aquatic vegetation. The lessons to be learned from the Chesapeake Bay experience and other areas of the world are: (1) the degradation of water quality and increase in oxygen stress occurred over decades; (2) multi-level and multi-institutional support is required to institute nutrient management schemes; (3) many years will pass before the results of implementation of nutrient controls will be seen in nutrient concentrations of aquatic systems, including the coastal ecosystem; (4) biological restoration will also require a long time to respond to the changes; (5) natural variability in climate may mask restoration because of fluctuations in freshwater discharge and the nonpoint sources of nutrients carried with it; (6) restoration of ecosystems following nutrient abatement is achievable; and (7) benefits accrue to multiple facets of society.

Coastal ecosystems have inherent large interannual variability, so that management actions that reduce nitrogen and phosphorus loading may not lead to immediate recovery toward the unenriched state. Riisgård et al. (1996) noted for the Kertinge Nor in Denmark that as external nutrient loading to the system was reduced significantly in recent years, nutrient release from the sediments has become important in determining the biological structure of the system. Richardson (1996) noted that interannual environmental variability was considerable for Danish coastal waters, including year-to-year variability in oxygen concentrations. Thus, decades of data may be necessary to statistically demonstrate that remedial actions have helped recovery of oxygen concentrations (Richardson 1996).

This view is most likely realistic, and is transferable to the northern Gulf of Mexico. Although eutrophication and hypoxia can be reversed by decreasing inputs of nutrients, the rates of recovery will be highly variable, the eutrophic state may be persistent, and recovery may be slow (Carpenter et al. 1998). The number of years required to obtain statistical evidence for a change in an ecosystem will vary and will be subject to inherently unstable biological systems and extreme events (e.g., weather conditions, algal blooms) that can and do change the direction of the system's development. The science of cultural eutrophication is past its infancy, but the science of restoration of enriched systems is not. Both need further refinement to identify and recommend courses of action to remediate anthropogenic additions of nutrients to coastal waters.

CHAPTER 10

Research Needs

In the preparation of this synthesis, several ideas were not fully explored because of a lack of either data or time to do the necessary synthesis of available data. Where data were missing, there are useful avenues of research and model refinements that can supply them. Some issues cannot be resolved adequately without further data collection and analysis, or improved and different modeling approaches. These efforts will not happen soon or come to completion quickly.

Additional insights can be gleaned from further examination of existing data for empirical relationships between freshwater discharge, nutrient concentrations, nutrient ratios, and nutrient flux with either short- or long-term changes in:

- areas of bottom-water hypoxia in mid-summer and through the seasonal cycle for transect C, or station C6;
- the volume of hypoxia and oxygen-deficient waters in mid-summer, and through the seasonal cycle for transect C, or station C6*; and
- surrogates for surface-water productivity or bottom-water hypoxia in dated sediment cores.

Suitable multivariate techniques and multiple-regression analyses should be applied to a thorough compilation of hydrographic data and indicators of Mississippi River system change. Where appropriate, cross-correlation analyses should be conducted on changes in the marine system with constituents of Mississippi River discharge.

There can be no determination of a marine system response to any nutrient changes within the Mississippi River system without the continued acquisition of basic hydrographic, chemical, and biological data related to the development and maintenance of hypoxia over seasonal cycles. Continued monitoring is an absolute must if we are to learn anything about restoration efforts. The types of monitoring that are currently in place should be continued for continuation of the long-term data set so that relationships with changing river constituents and loads can be determined. The data acquisition must be consistent with that already collected and must be continuous. The triad of mid-summer shelf-wide hypoxia surveys, monthly transects off Terrebonne Bay, and an instrumented array at a station in the core of the hypoxic zone off Terrebonne Bay should be continued for an optimal combination of spatial and temporal scales of measurement. A better definition of the shelf-wide extent of hypoxia would necessitate additional shelf-wide cruises during the year.

Understanding the physical structure of the water column is relevant to the study of dissolved oxygen cycling. Likewise, physical measurements alone do not provide the information on the biological and chemical processes leading to the development and maintenance of hypoxia. An integrated and coordinated survey program is essential for the documentation of oxygen concentrations and the extent of shelf hypoxia and the relevance of physical and biological processes in its formation and maintenance. Finer-scale resolution of vertical mixing of oxygen in relation to winds and horizontal advection in relation to wind-induced currents are important components of the physical environment that with better understanding will help define the timing and location of hypoxia. Multiple-instrumented arrays with instruments in the vertical and spatially oriented cross-shelf and along-shelf directions will help better define the temporal and spatial development of hypoxia.

Process-oriented experiments should be conducted to better define the processing of nutrients within the Mississippi River plume and in the extended plume over the broad area where hypoxia is most likely to develop. Much remains unknown concerning the flux of nutrients and their relative proportions in relation to productivity, the types of phytoplankton communities that develop, trophic consequences, and the subsequent flux of various carbon constituents that result in the depletion of oxygen. Experiments to determine the timing and relative contribution of various sources of carbon to the formation and maintenance of hypoxia over several annual cycles should be conducted at several sites along the Louisiana coast.

The fate of fluxed carbon to the seabed within the zone of hypoxia is not known. Whether the carbon is buried, remineralized, resuspended, and transported from the area, or incorporated into benthic macrofaunal biomass and benthic food webs, or transformed by combinations of these processes is unknown. Many potential interactions within the seabed for nutrient cycling and carbon processing have the potential for both positive and negative feedback to the eutrophication of the Louisiana shelf. These numerous processes, pelagic–benthic coupling, and aspects of external forcing, such as nutrient flux and ratios, are best studied at a more restricted number of locations rather than a strategy that favors spatial coverage at the expense of the necessary temporal coverage to define the natural processes within a broader context of climatic variability.

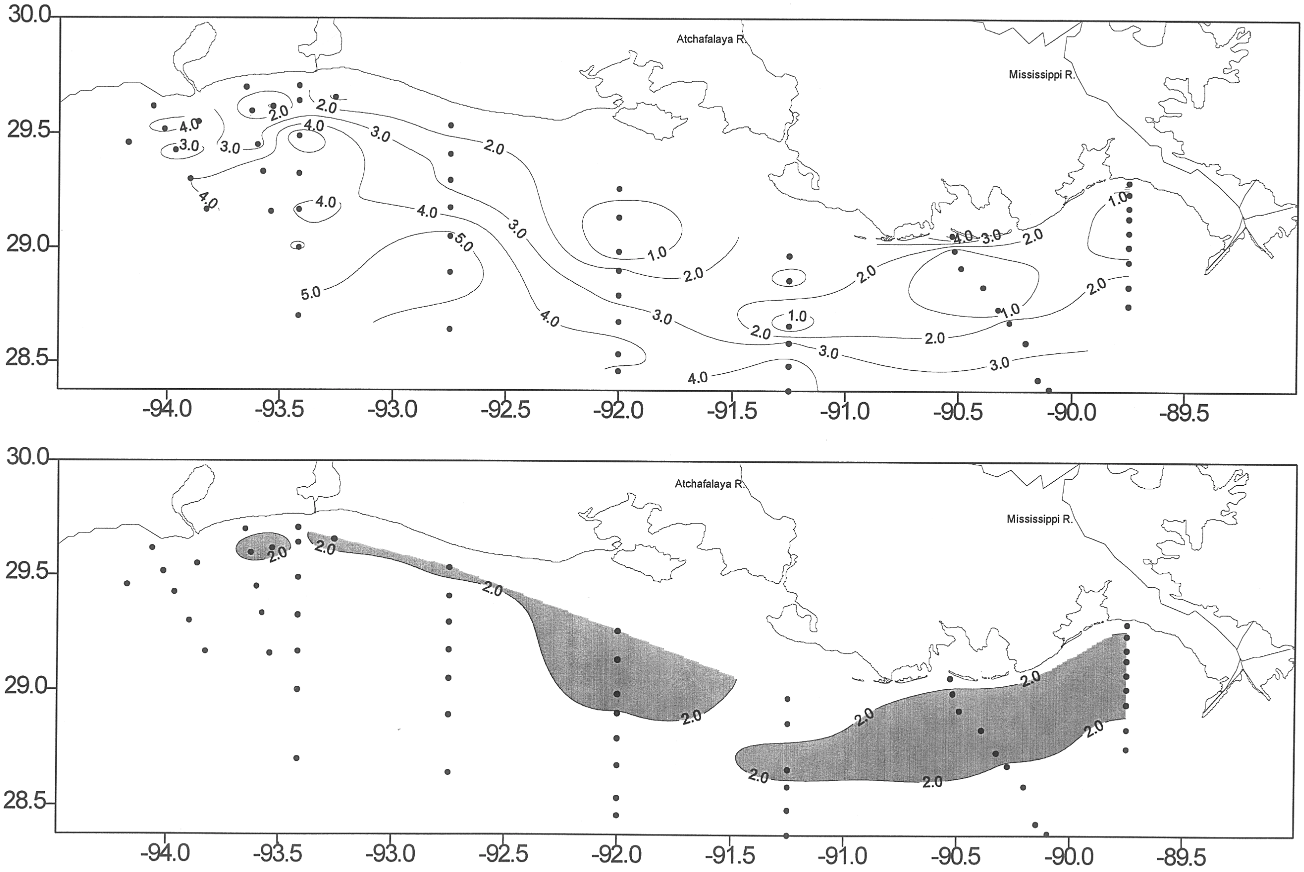
Much progress has been made in determining long-term ecosystem responses by examination of surrogates of surface-water productivity and bottom-water hypoxia in dated sediment cores. These studies should be continued over the long term as comparative measurements of ecosystem response to changes in the Mississippi River system. The spatial resolution of these retrospective analyses can be increased. An attempt should be made to identify areas under the influence of the Atchafalaya River plume for similar studies, although the presence of accumulating sediments is not likely, except in some bathymetric low areas. Additional surrogates for eutrophication and hypoxia should be added to those components already studied.

The process of coastal eutrophication in the northern Gulf of Mexico at the terminus of the Mississippi River system has taken many decades. While it is likely that remedial actions can alleviate the hypoxia problem in the Gulf, the marine ecosystem response may be slow, and changes will be masked by the natural system’s high variability. Decades of data may be necessary to statistically demonstrate that remedial actions have helped recovery of oxygen concentrations.

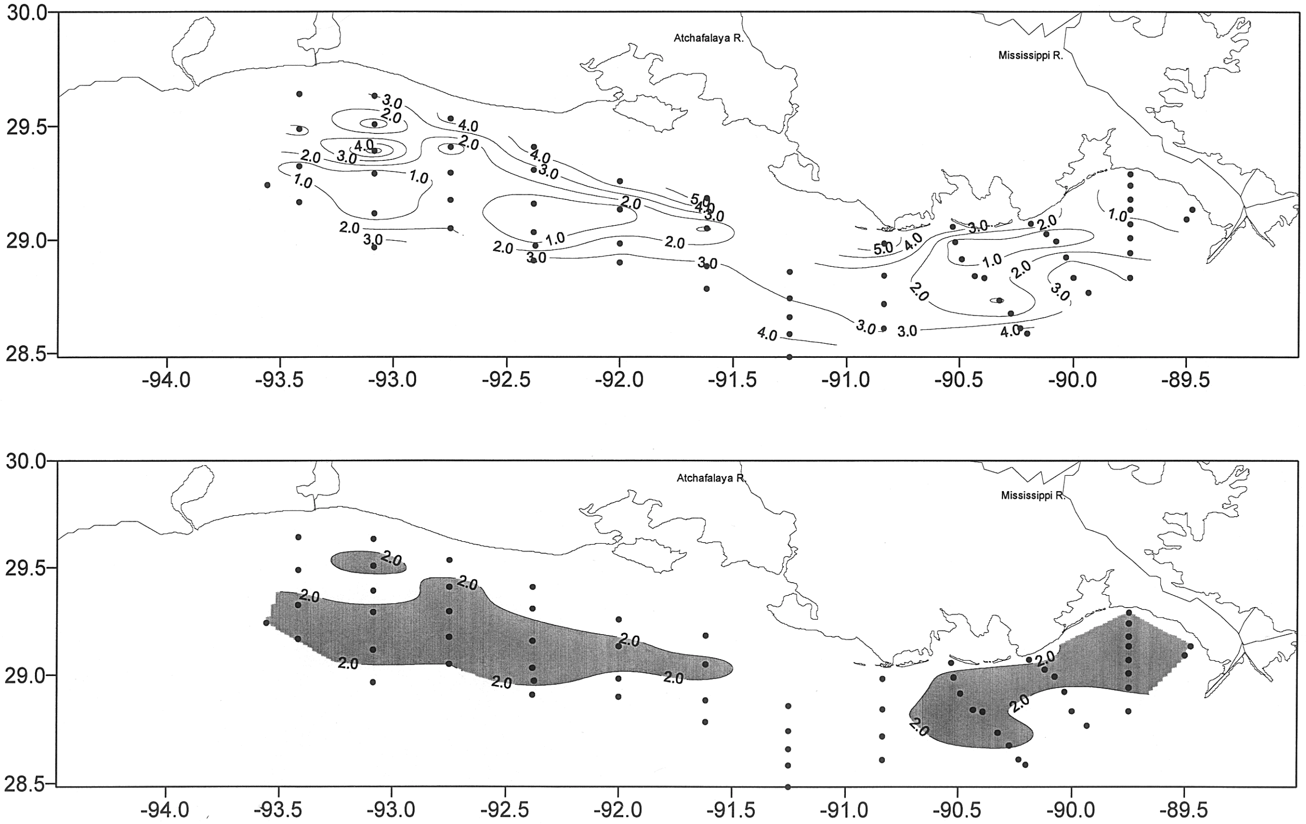
Appendix A

The graphics in this appendix display the results from 1985–98 mid-summer shelf-wide surveys, oxygen isopleths, and area (stippled) of near-bottom hypoxia. The dots in the graphics represent the stations sampled. The contours were generated by Surfer^{TM} graphics, and the results are derived from unpublished data of Rabalais, Turner, and Wiseman.

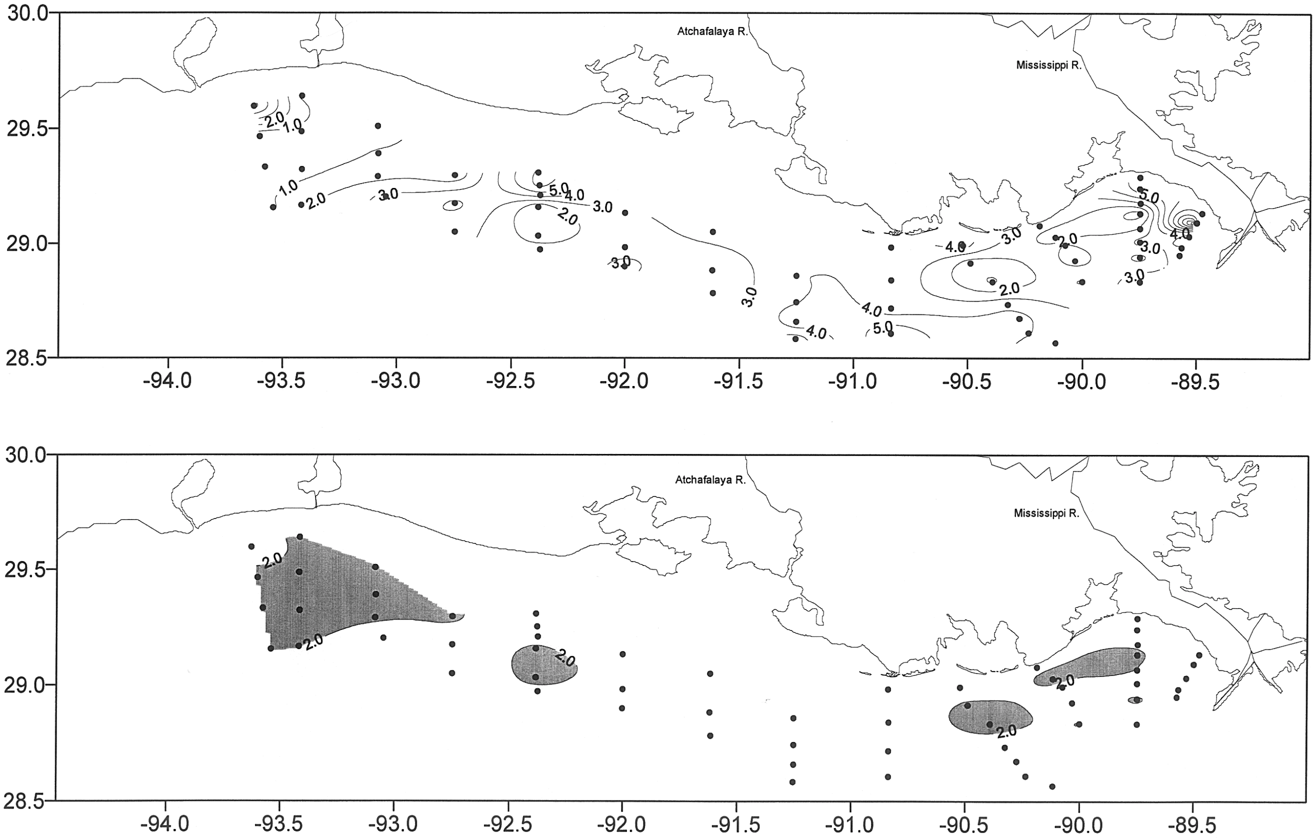
Bottom Dissolved Oxygen (mg/L)
July 15 - 20 1985



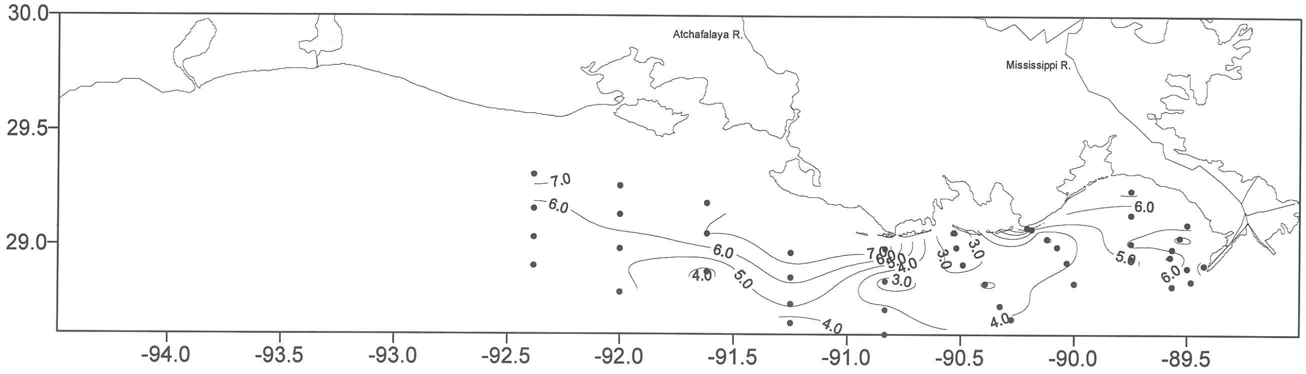
Bottom Dissolved Oxygen (mg/L)
July 7 - 18 1986

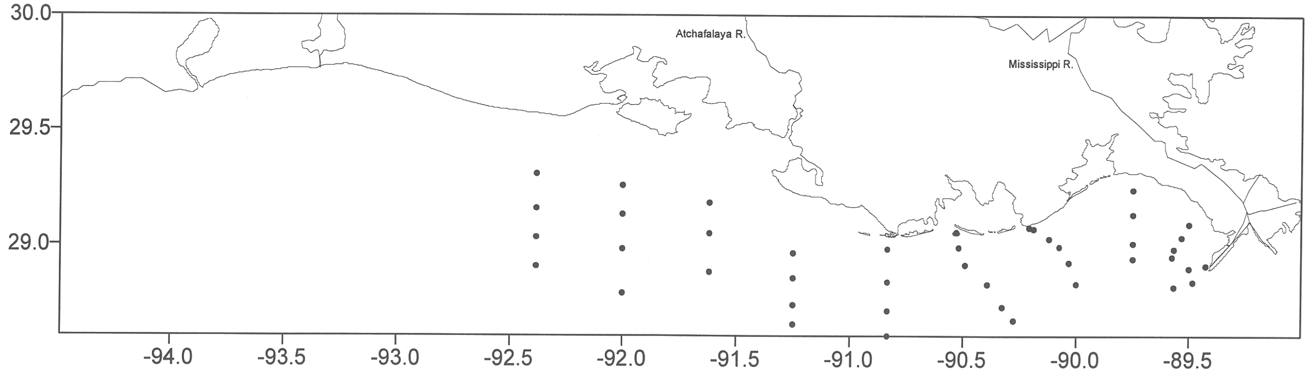


Bottom Dissolved Oxygen (mg/L)
July 1 - 5 1987

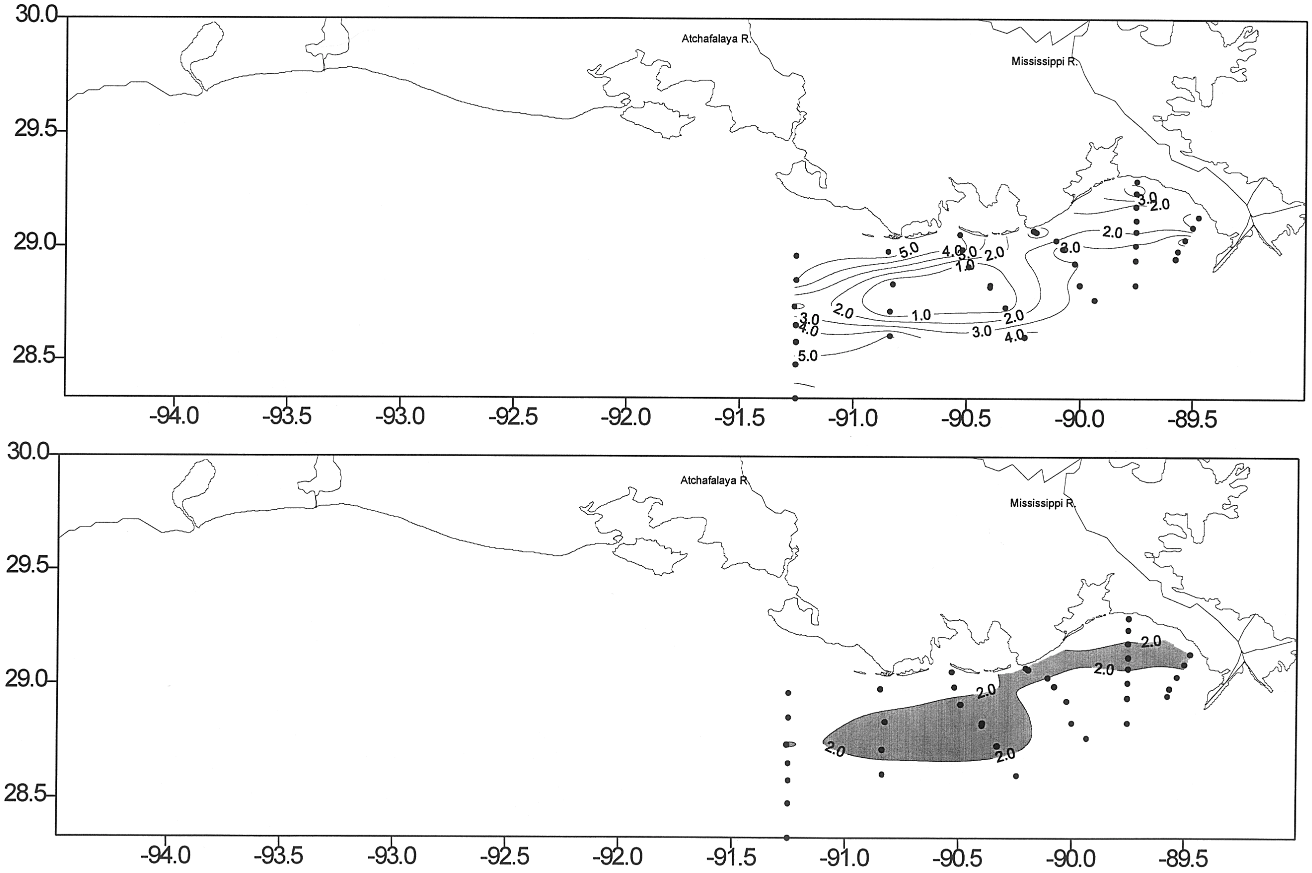


Bottom Dissolved Oxygen
August 12 - 16 1988

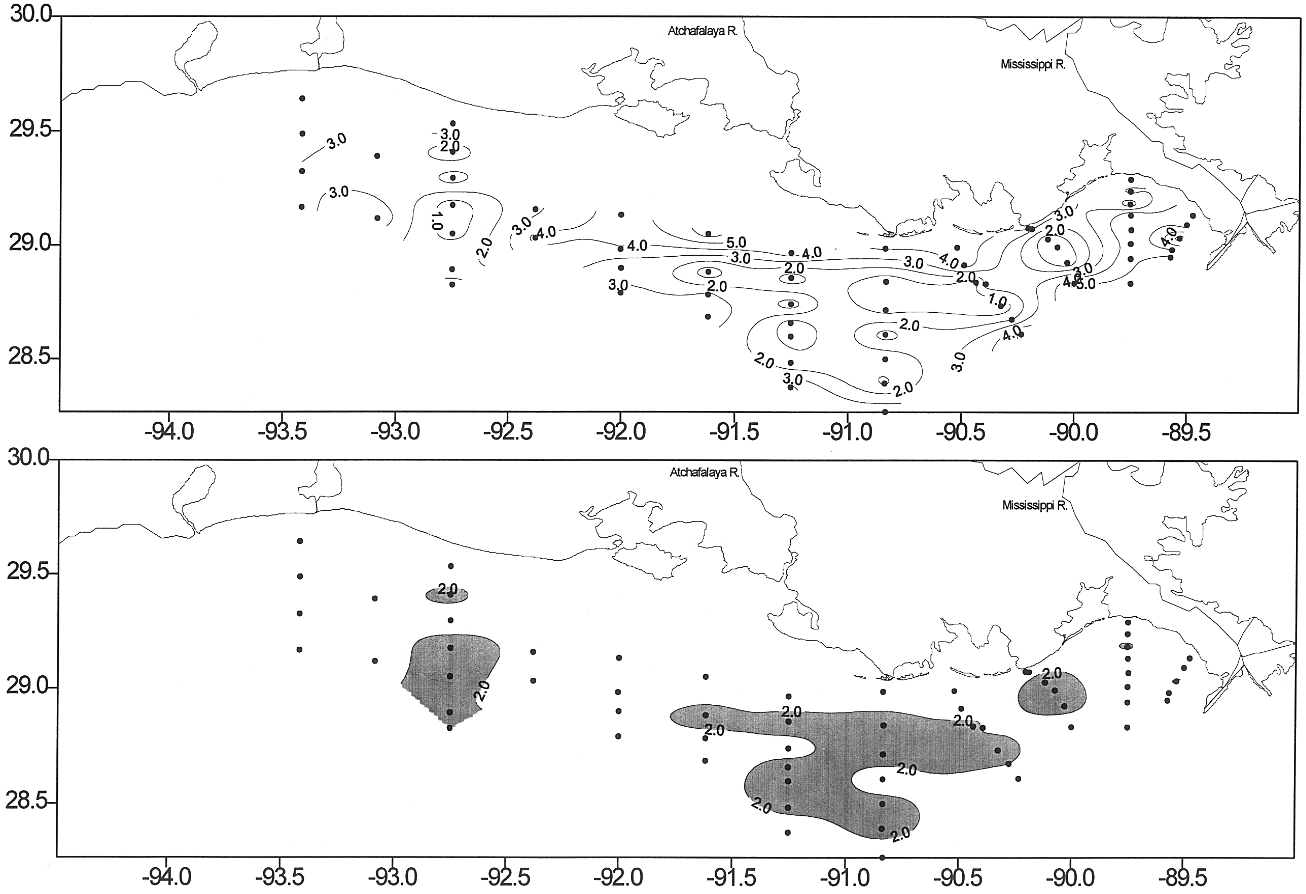


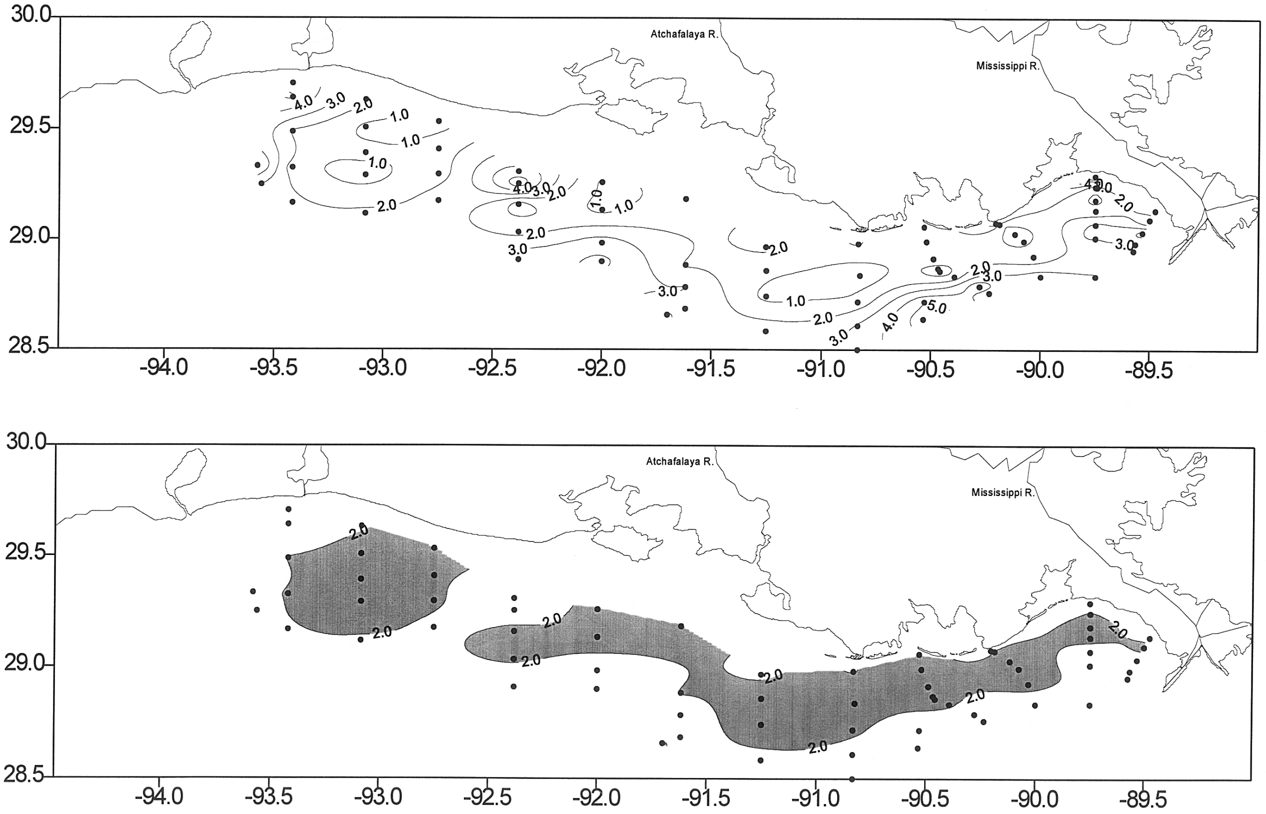


Bottom Dissolved Oxygen (mg/L)
August 4 - 10 1989



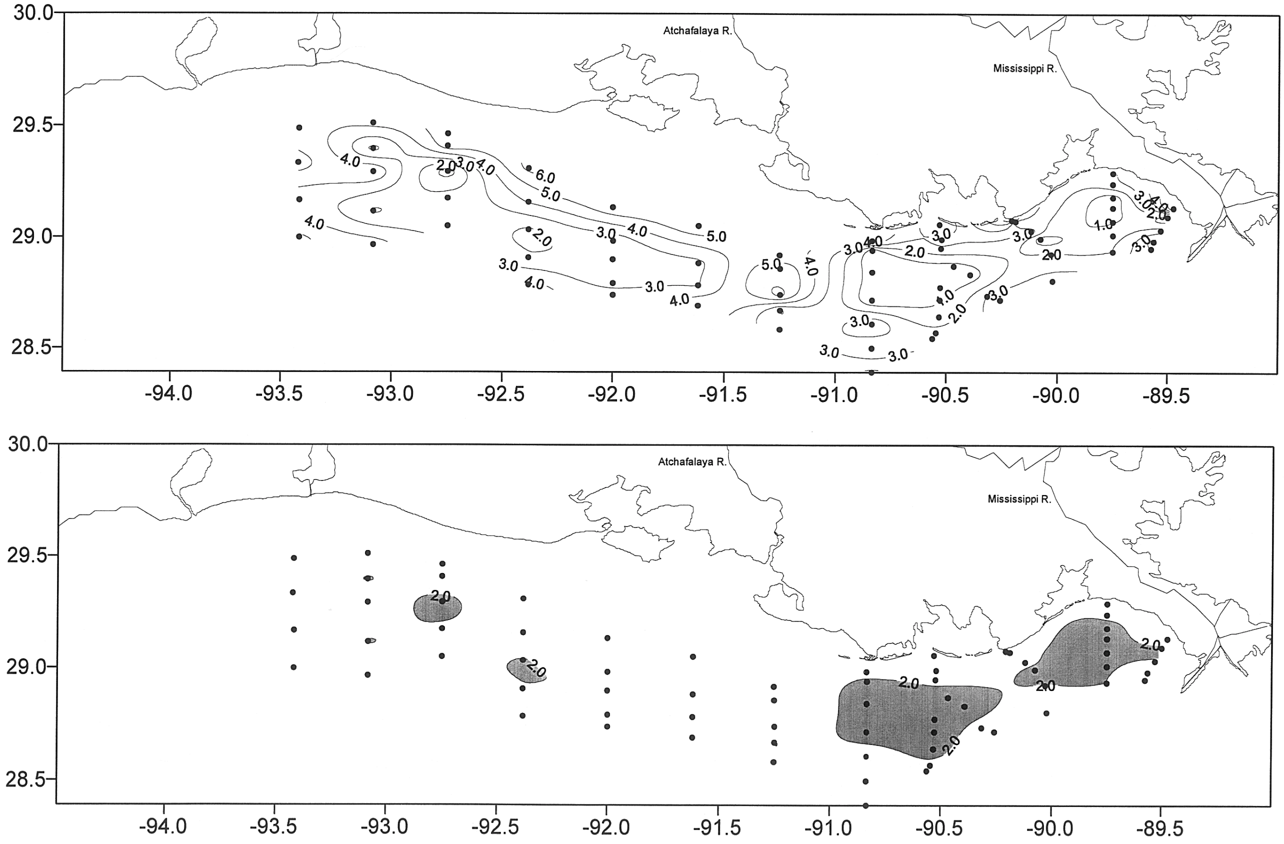
Bottom Dissolved Oxygen (mg/L)
July 23 - 27 1990



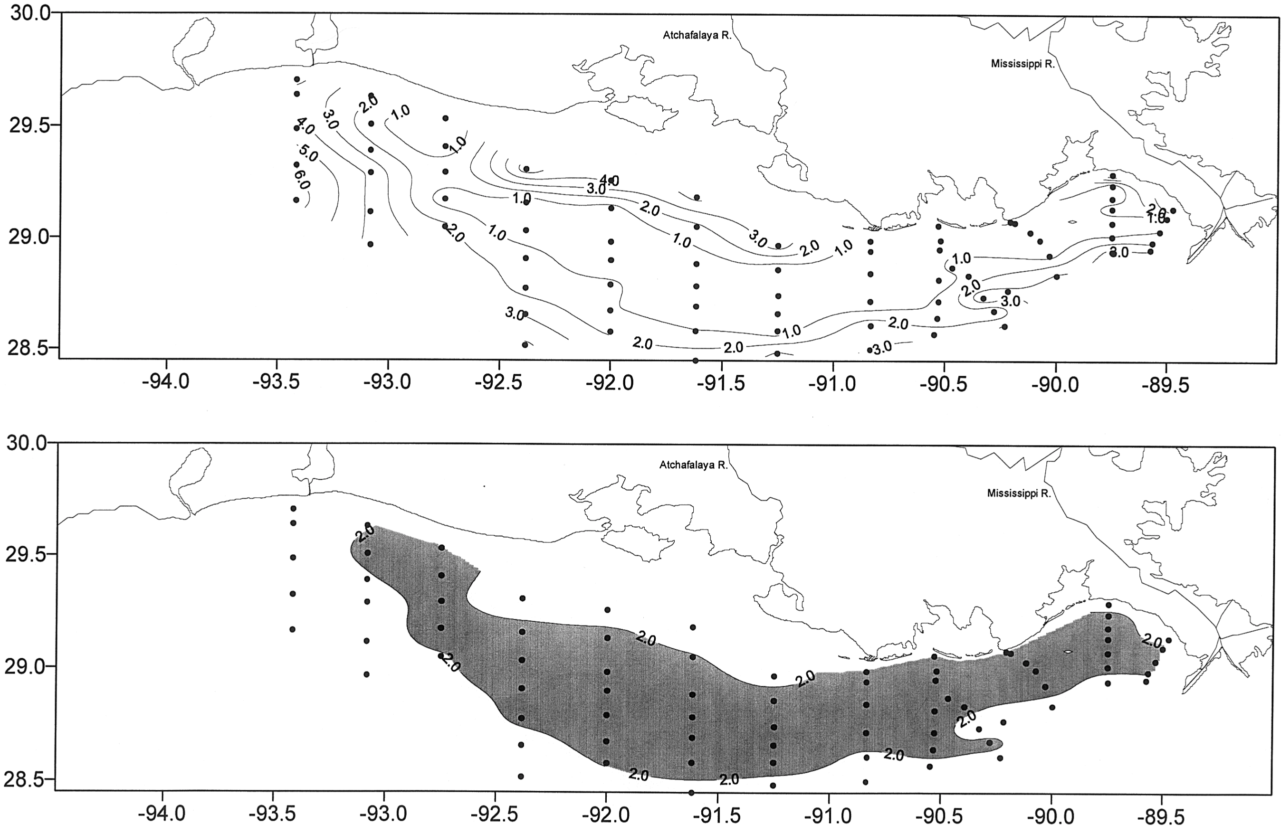


Bottom Dissolved Oxygen (mg/L)
July 16 - 20 1991

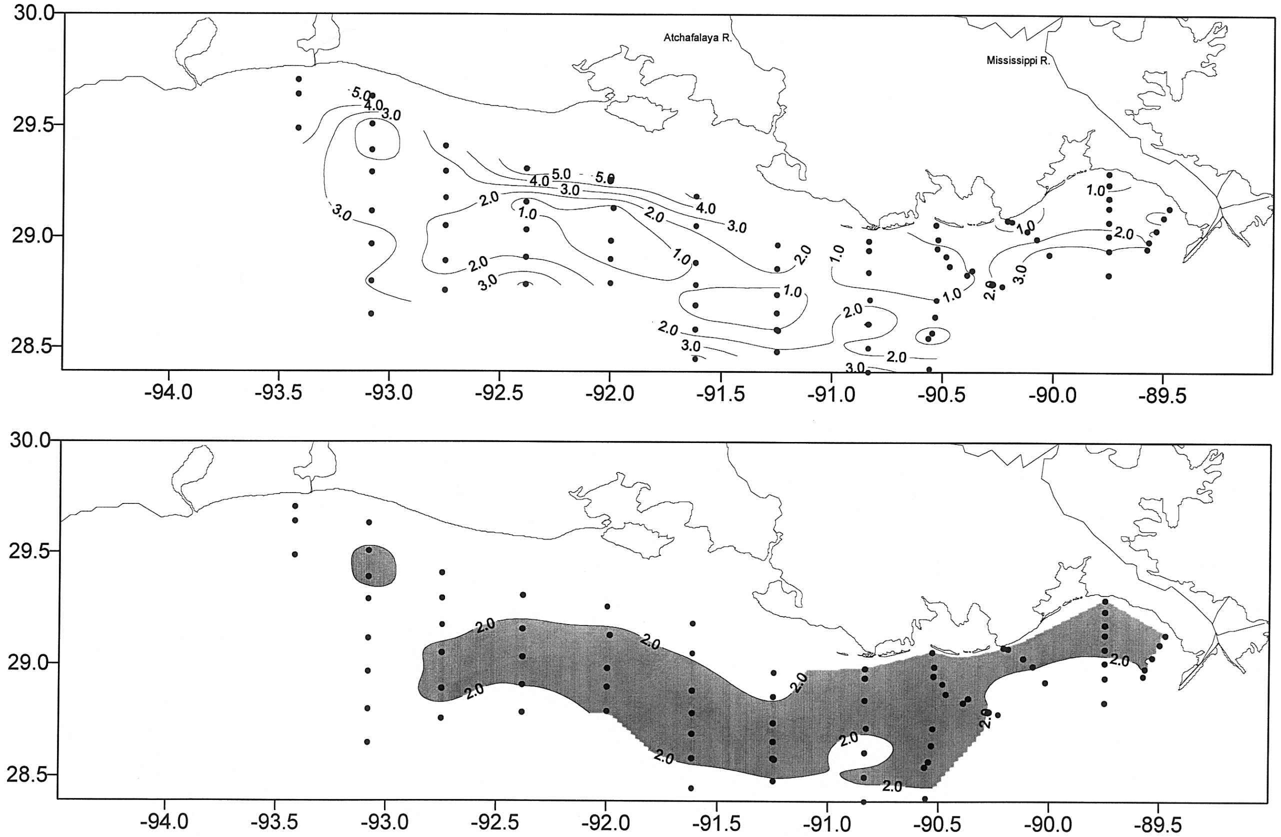
Bottom Dissolved Oxygen (mg/L)
July 24 - 29 1992

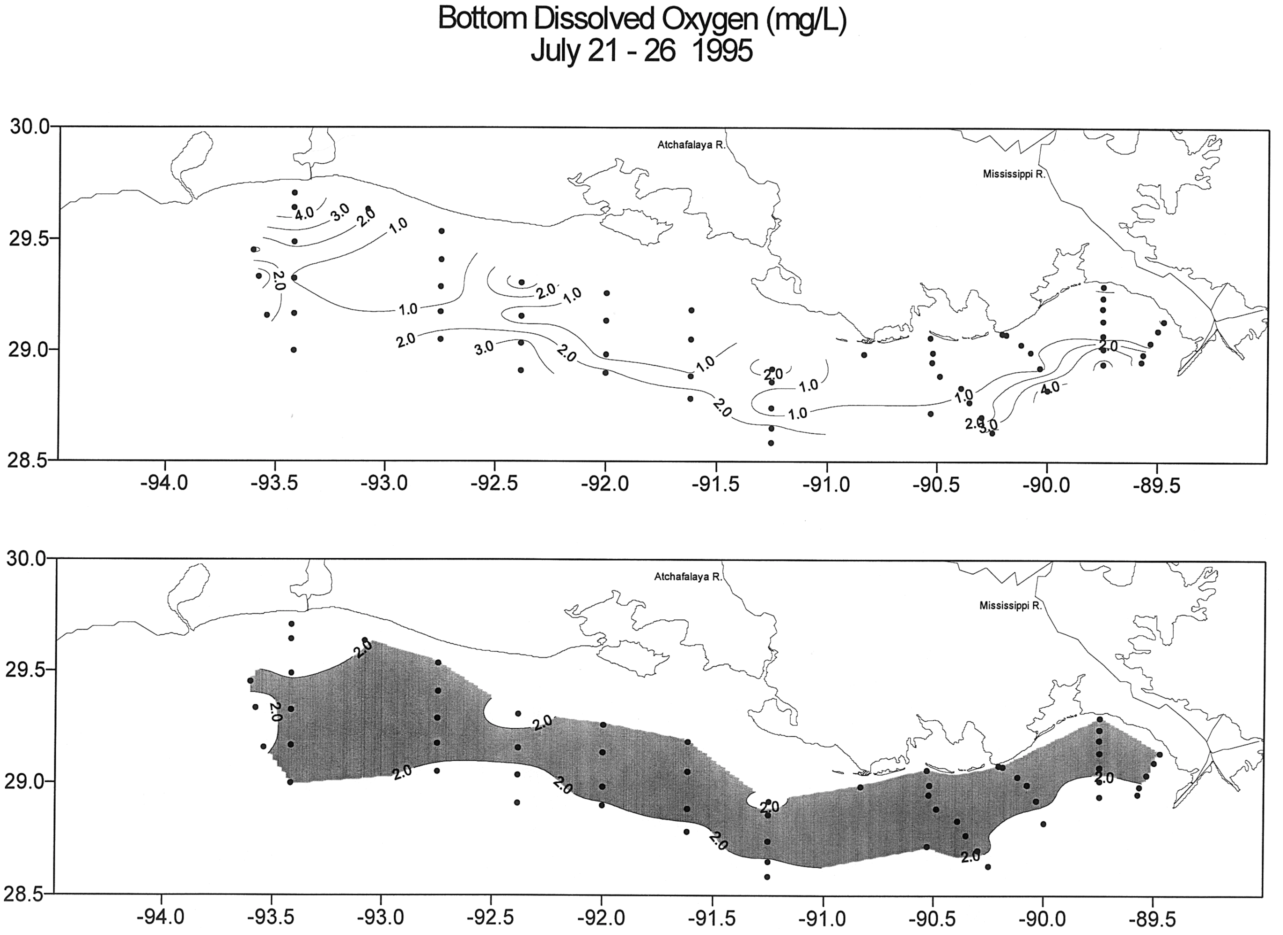


Bottom Dissolved Oxygen (mg/L)
July 24 - 30 1993

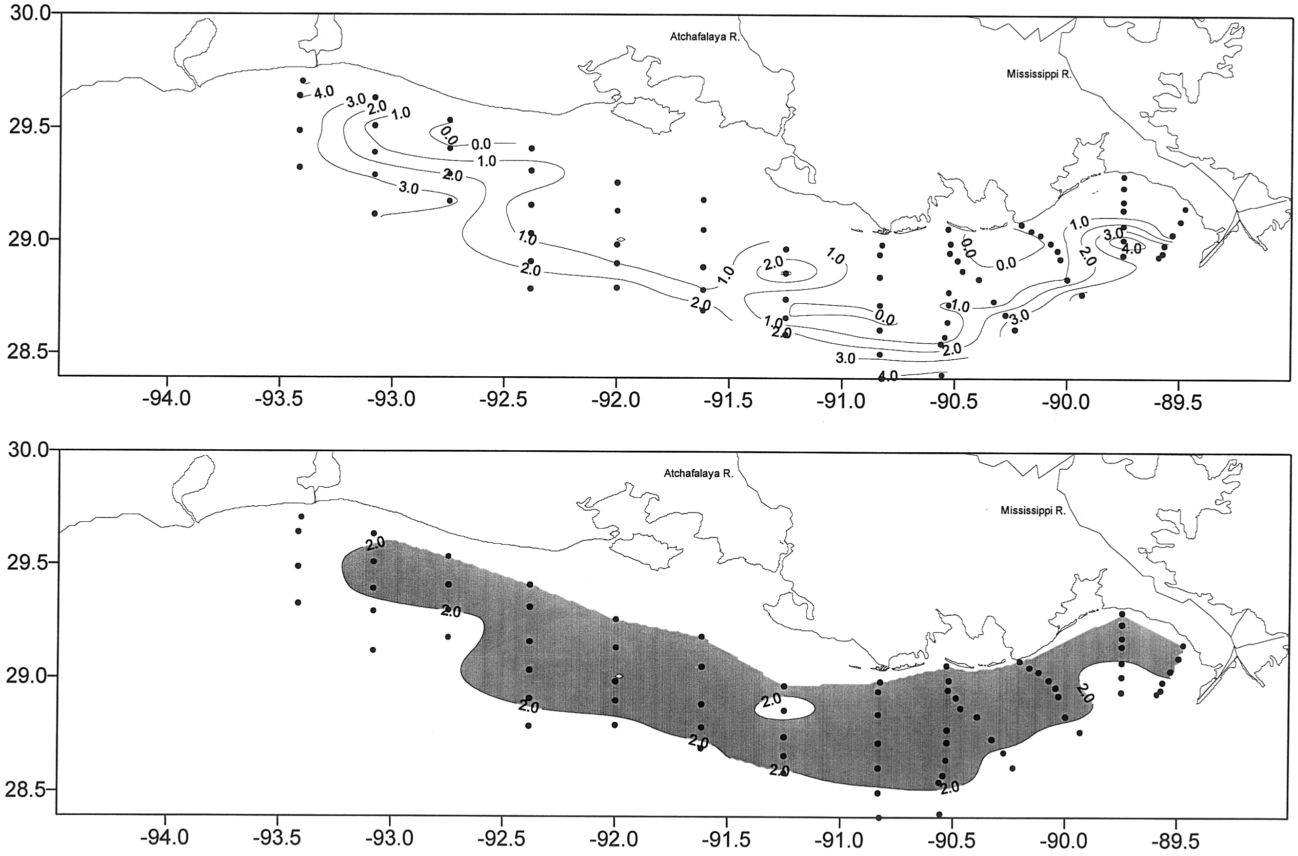


Bottom Dissolved Oxygen (mg/L)
July 24 - 29 1994

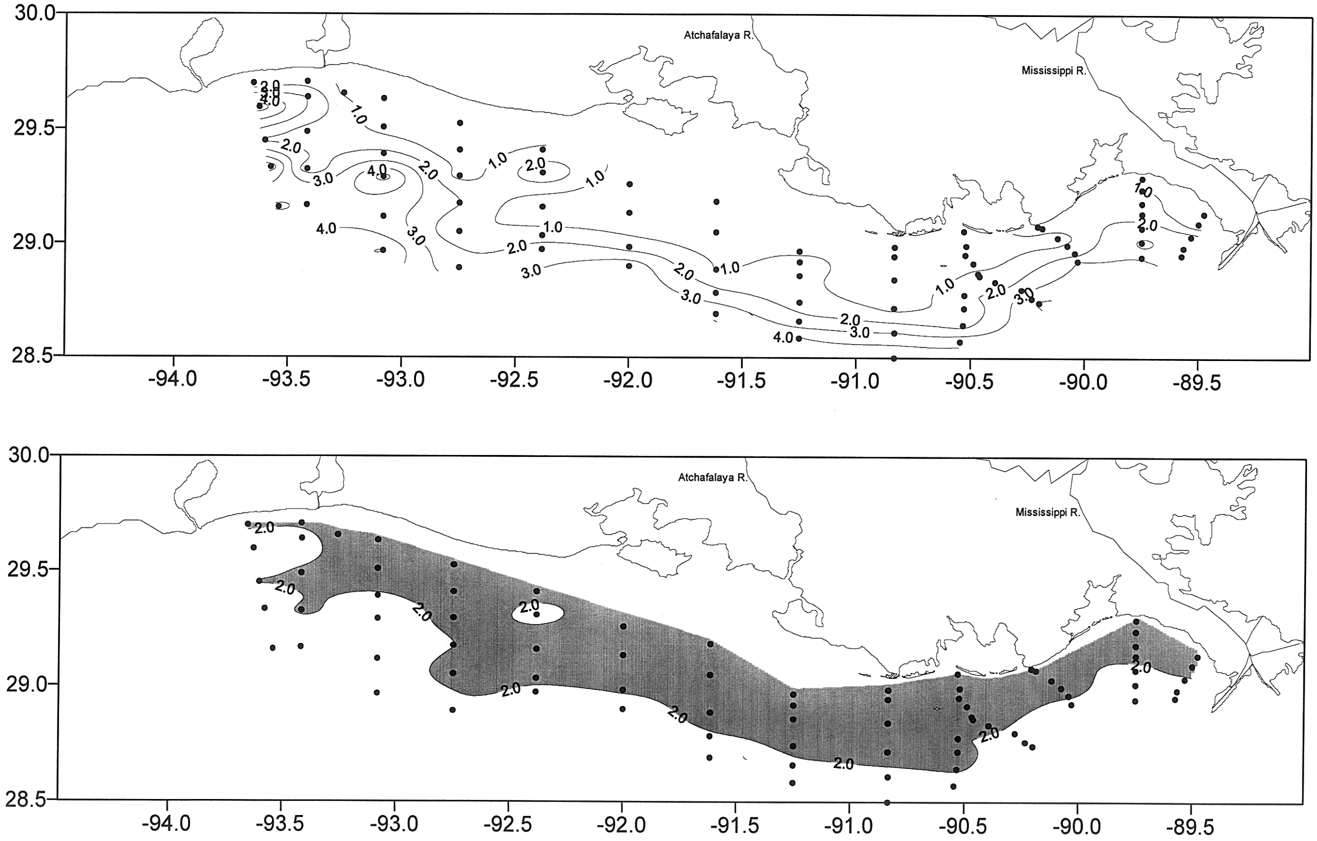



Bottom Dissolved Oxygen (mg/L)
July 21 - 26 1995
Atchafalaya R.
Mississippi R.
30.0
29.5
29.0
28.5
-94.0
-93.5
-93.0
-92.5
-92.0
-91.5
-91.0
-90.5
-90.0
-89.5
4.0
3.0
2.0
1.0

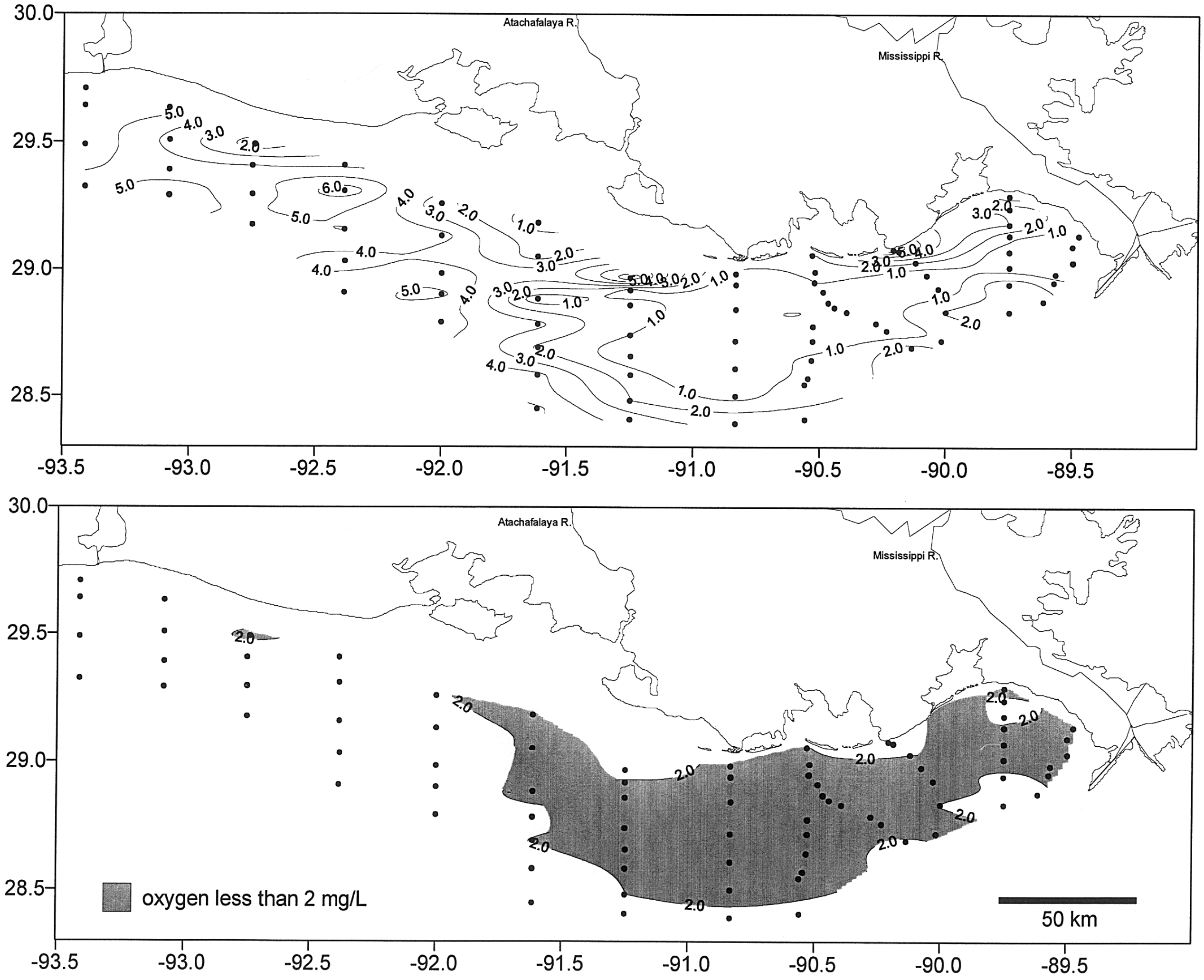
Bottom Dissolved Oxygen (mg/L)
July 23 - 28 1996



Bottom Dissolved Oxygen (mg/L)
July 23 - 28 1997



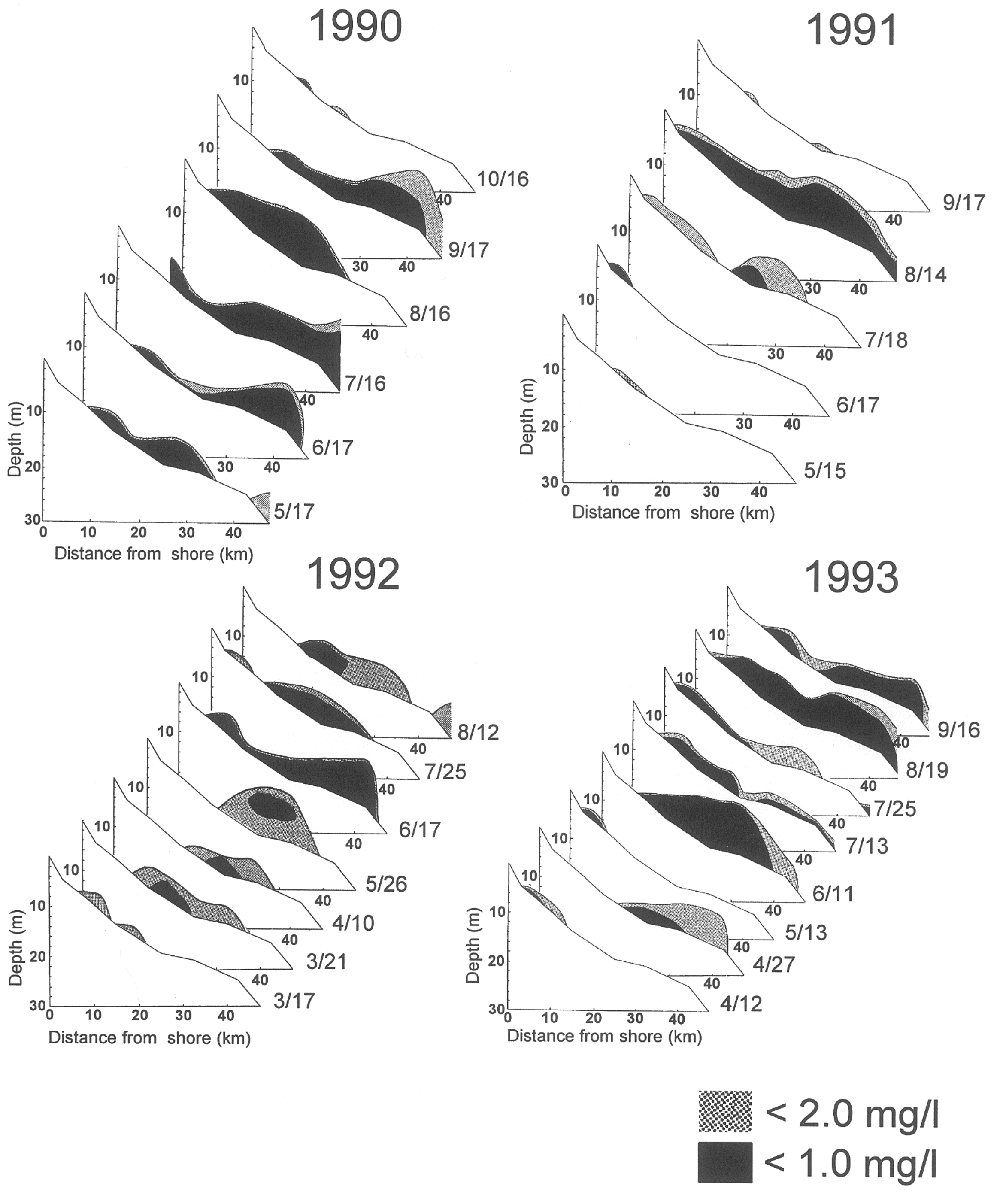
Bottom Dissolved Oxygen (mg/L)
July 21-25, 1998

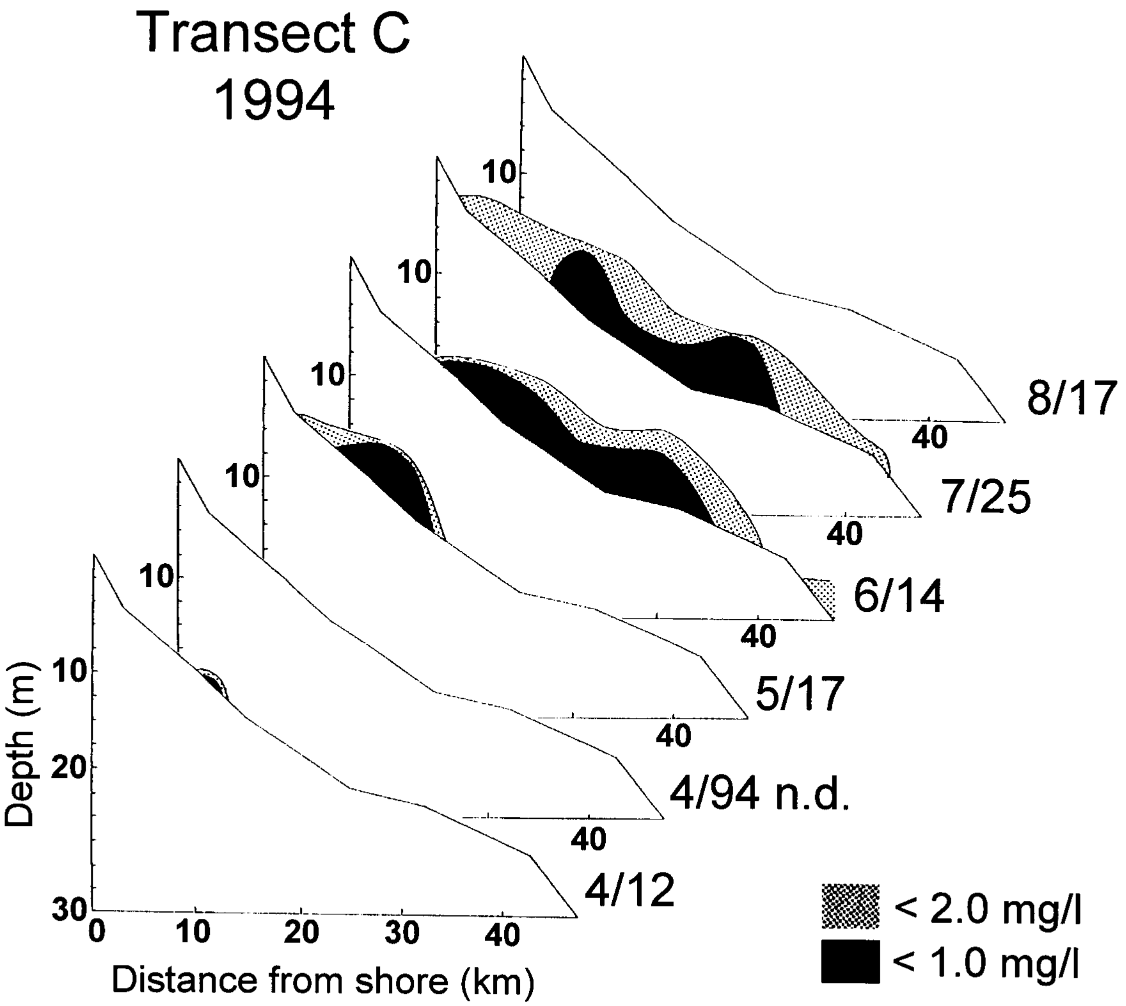


Appendix B

The graphics in this appendix display monthly cross-shelf contours of < 2 mg/l (stippled) and < 1 mg/l (black) (1990–94). Note that 1986 is shown in Figure 3.16 of this report. The data in the graphics are derived from unpublished data of Rabalais, Turner, and Wiseman.

Transect C

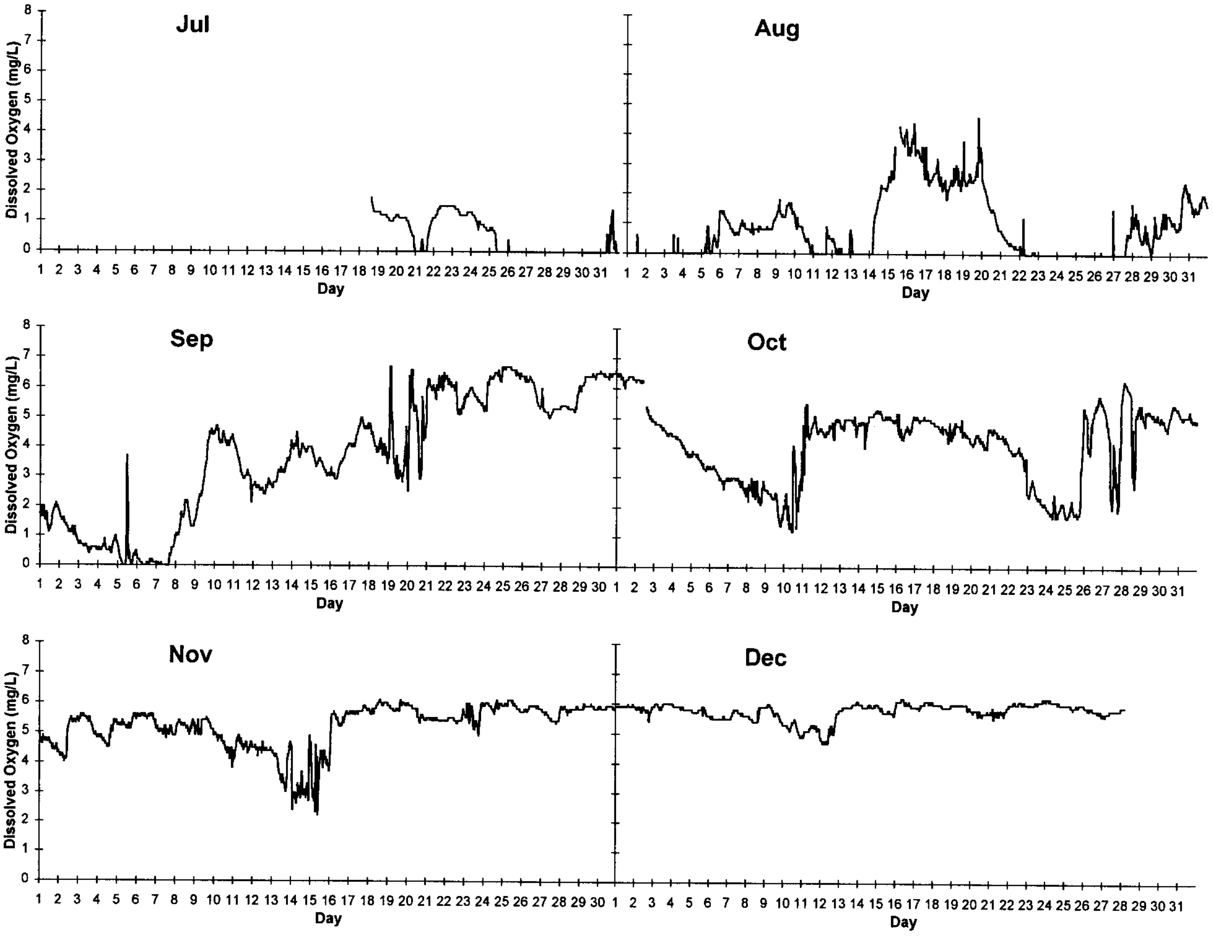

1990
10/16
9/17
8/16
7/16
6/17
5/17
Depth (m)
Distance from shore (km)
1991
9/17
8/14
7/18
6/17
5/15
Depth (m)
Distance from shore (km)
1992
8/12
7/25
6/17
5/26
4/10
3/21
3/17
Depth (m)
Distance from shore (km)
1993
9/16
8/19
7/25
7/13
6/11
5/13
4/27
4/12
Depth (m)
Distance from shore (km)
< 2.0 mg/l
< 1.0 mg/l


Transect C
1994
8/17
7/25
6/14
5/17
4/94 n.d.
4/12
10
40
Depth (m)
10
20
30
0
10
20
30
40
Distance from shore (km)
▨ < 2.0 mg/l
■ < 1.0 mg/l

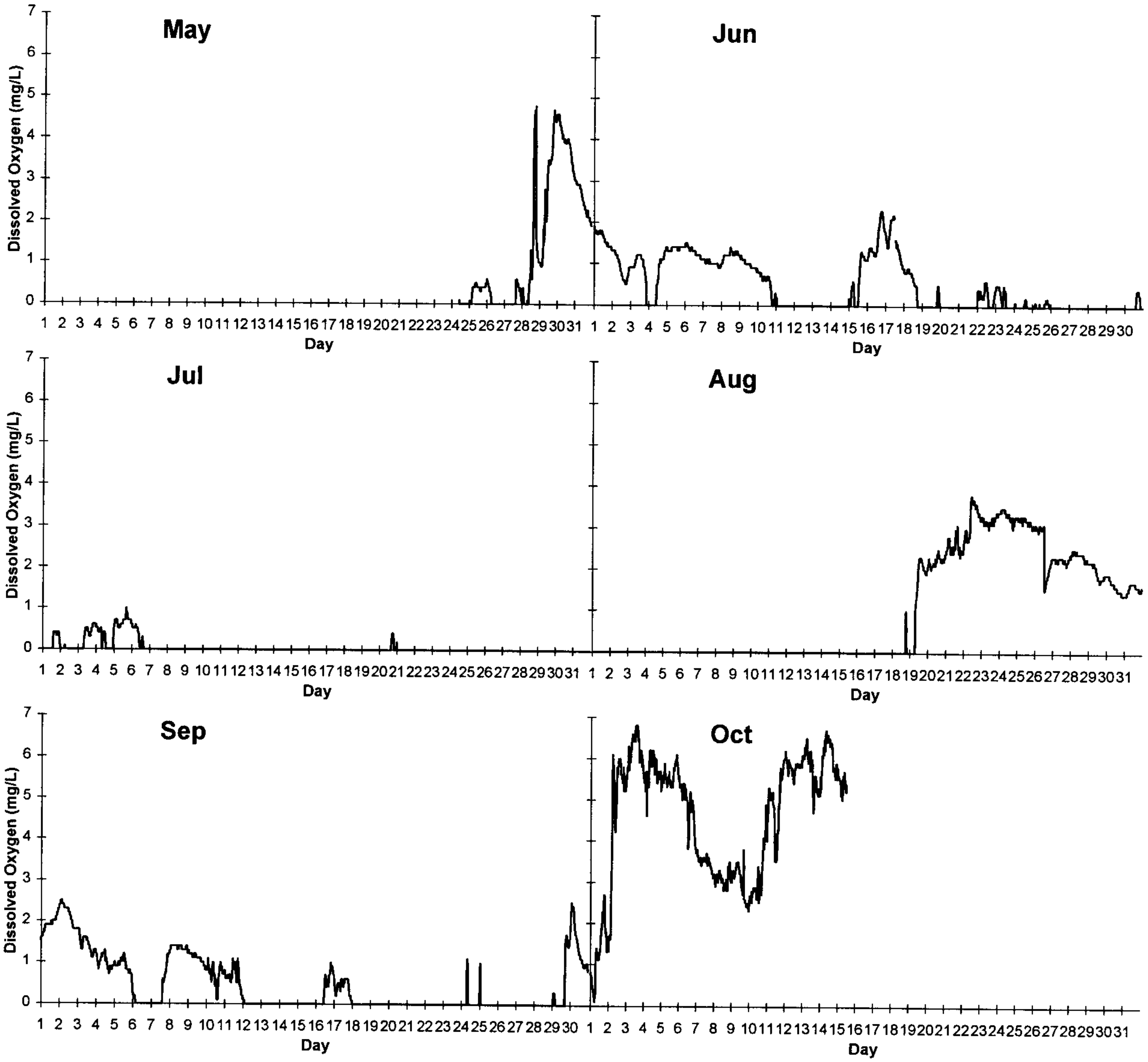
Appendix C

The graphics in this appendix display continuous oxygen data for station C6A or C6B (1989–97). Although measurements were made at 15-minute intervals, 1-hour intervals are depicted in this appendix. The results presented here are derived from unpublished data of Rabalais, Turner, and Wiseman.

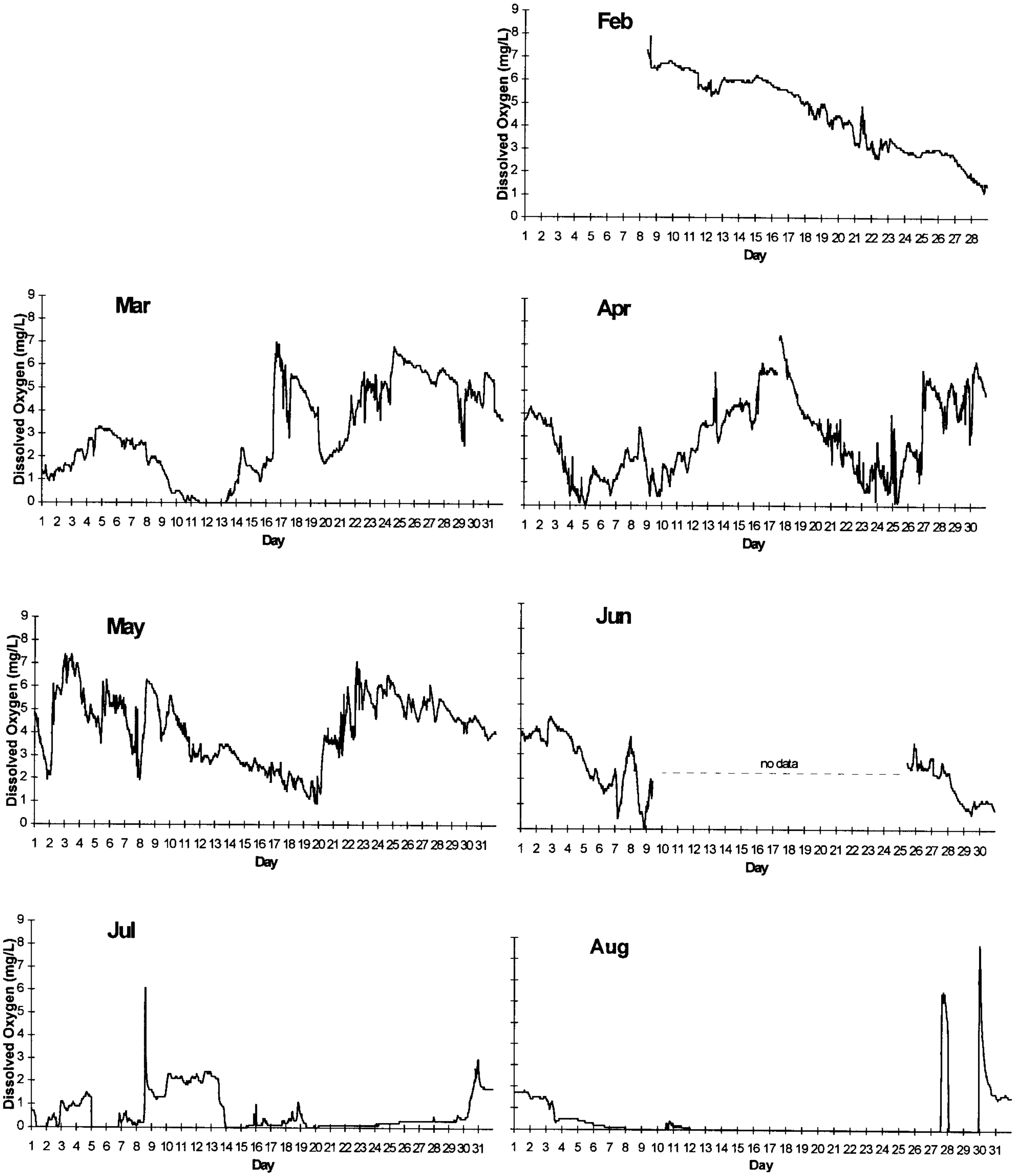
Station C6A 1989 Bottom Oxygen (mg/L)



Station C6A 1990 Bottom Oxygen (mg/L)

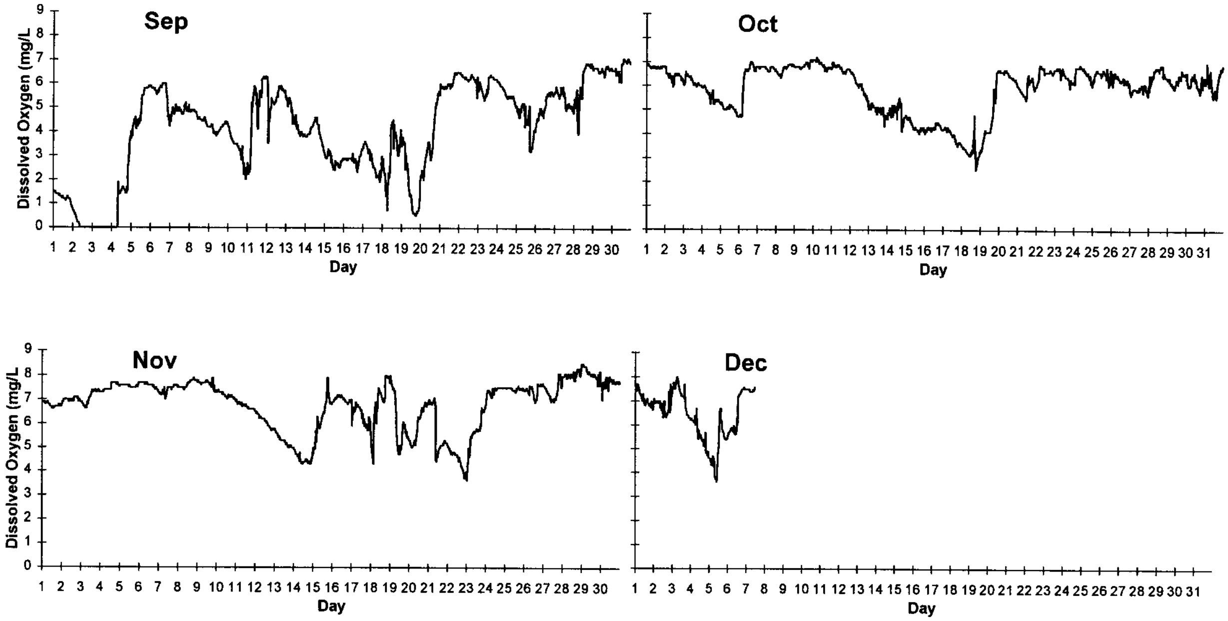


Station C6B 1991 Bottom Oxygen (mg/L)

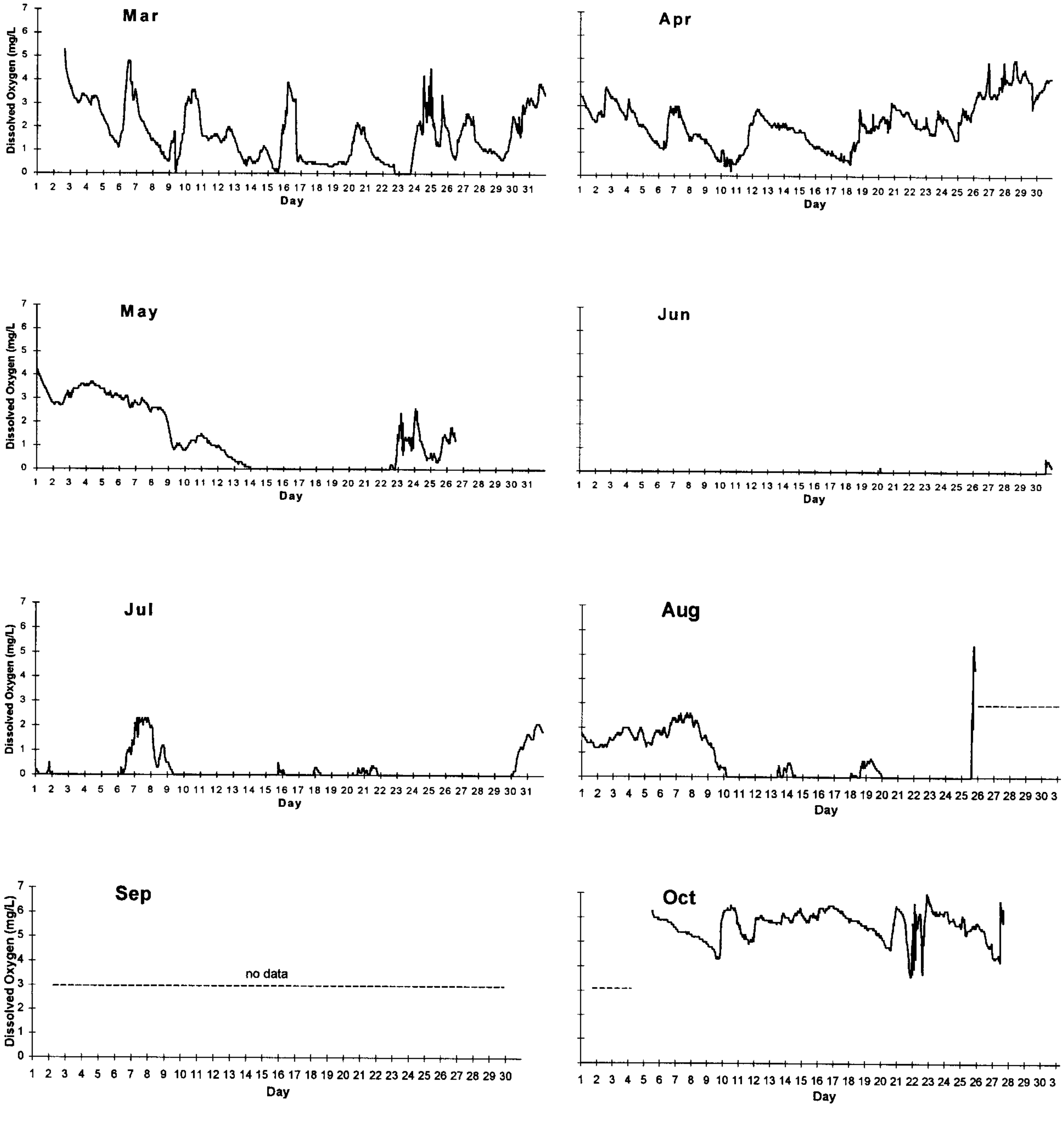


continued on next page

Station C6B 1991 Bottom Oxygen (mg/L) continued



Station C6B 1992 Bottom Oxygen (mg/L)



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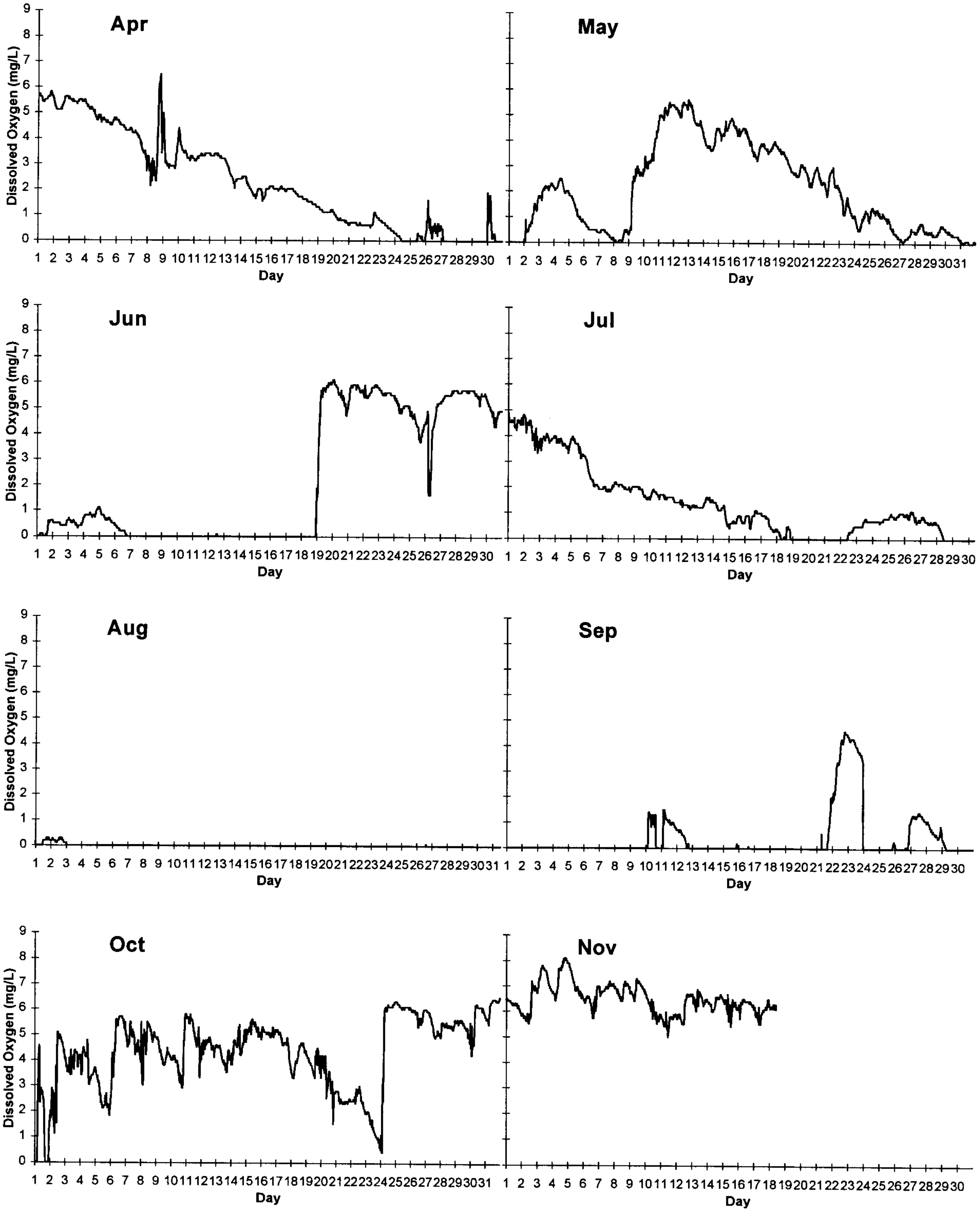

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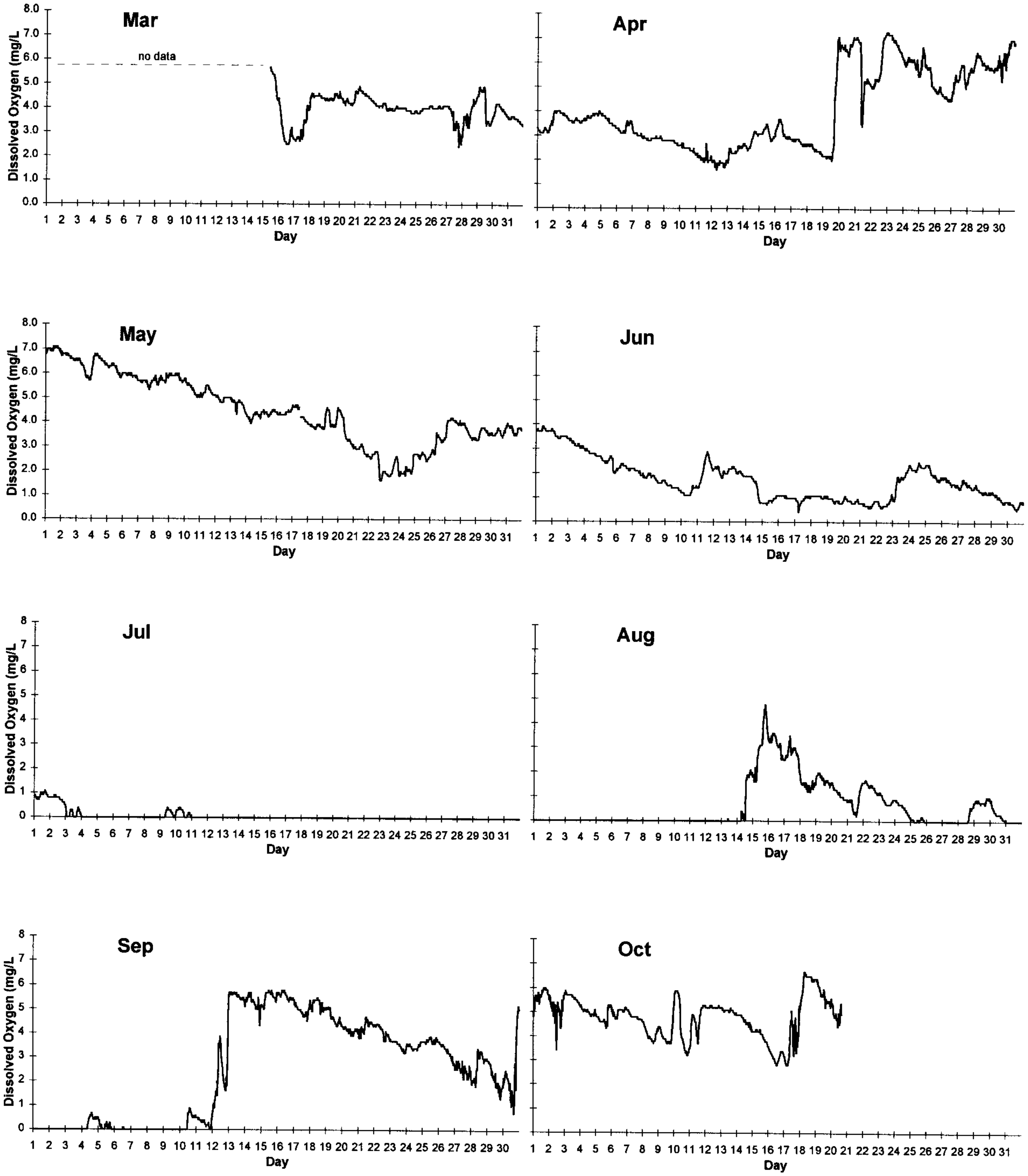

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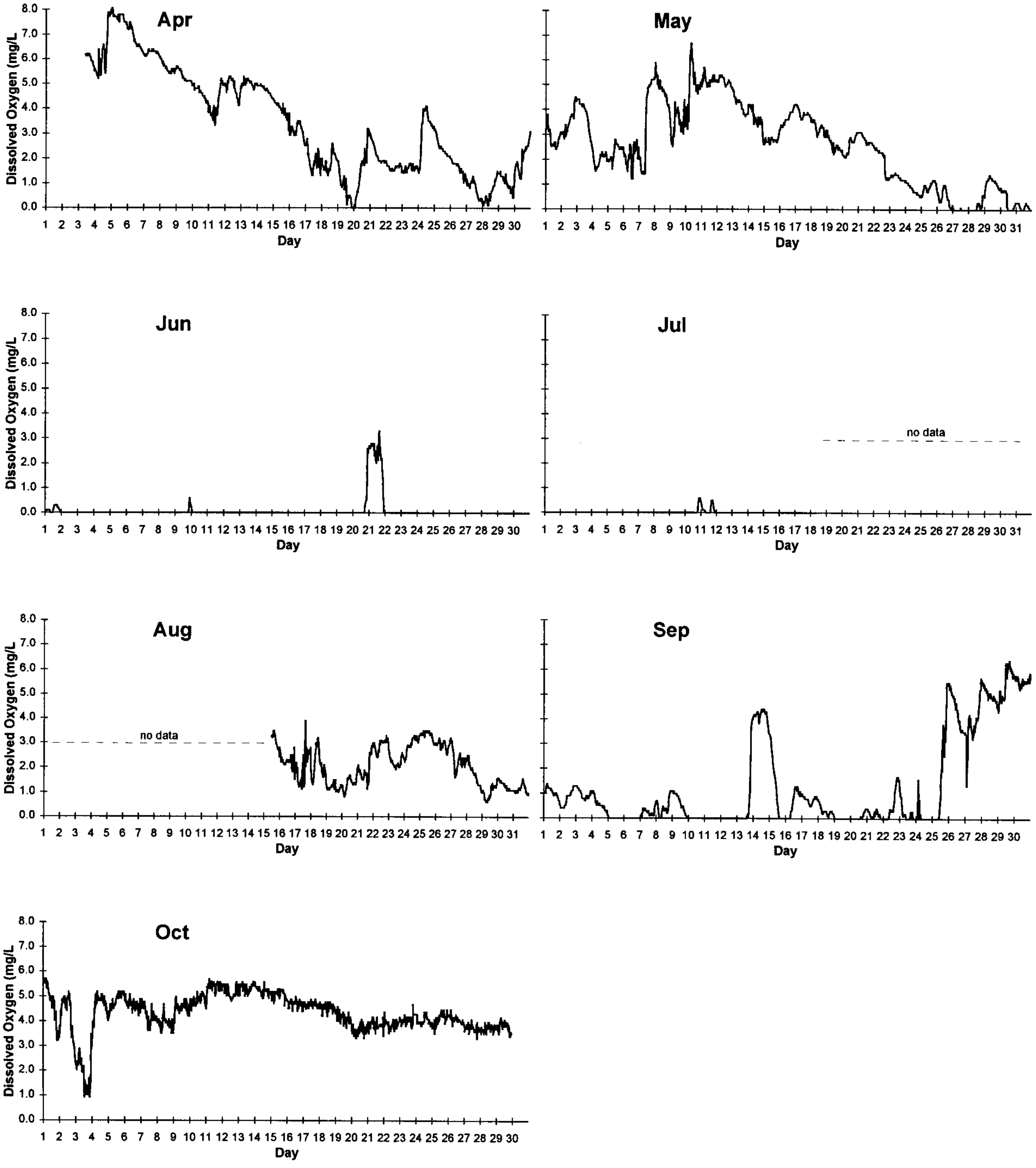

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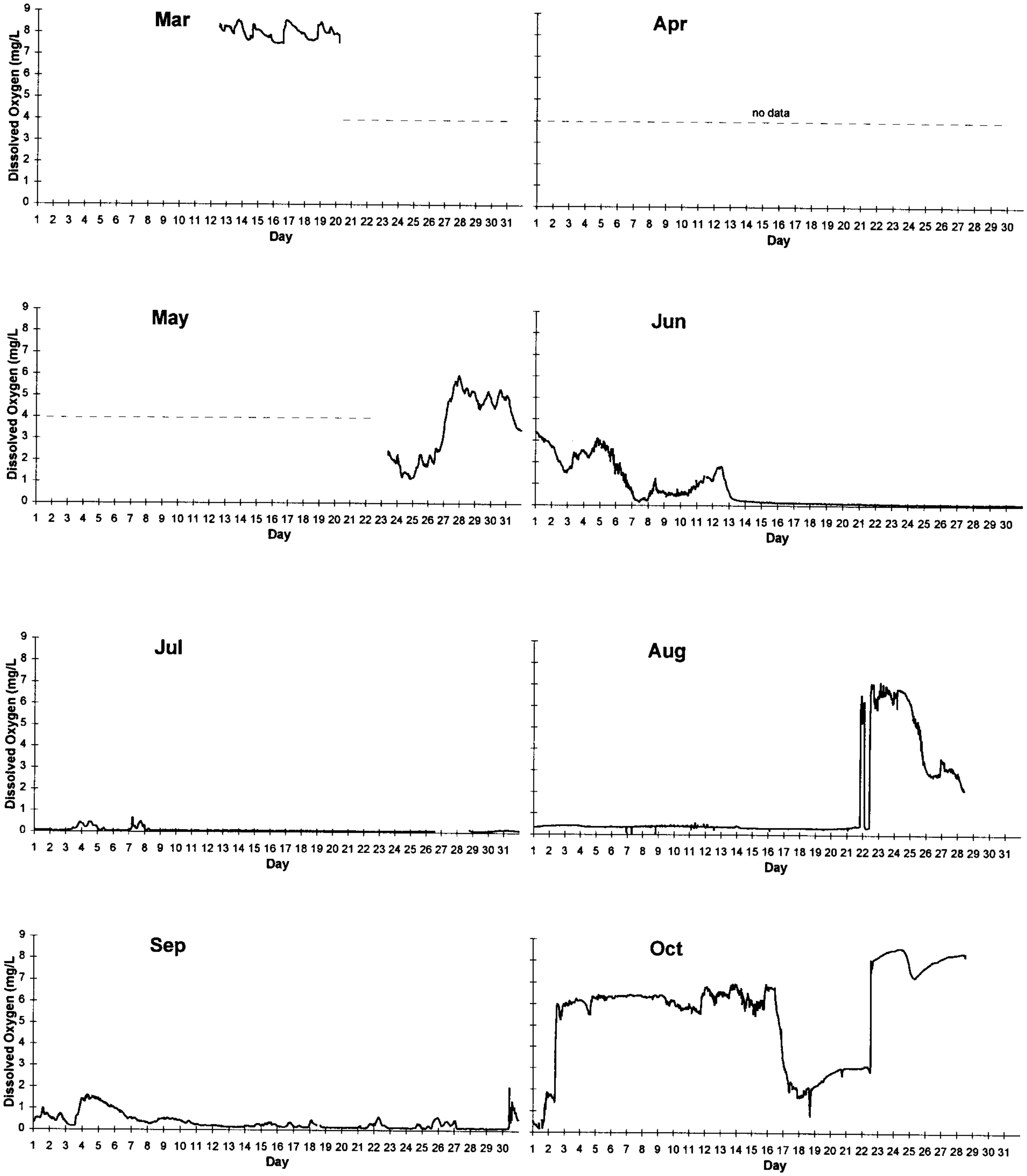

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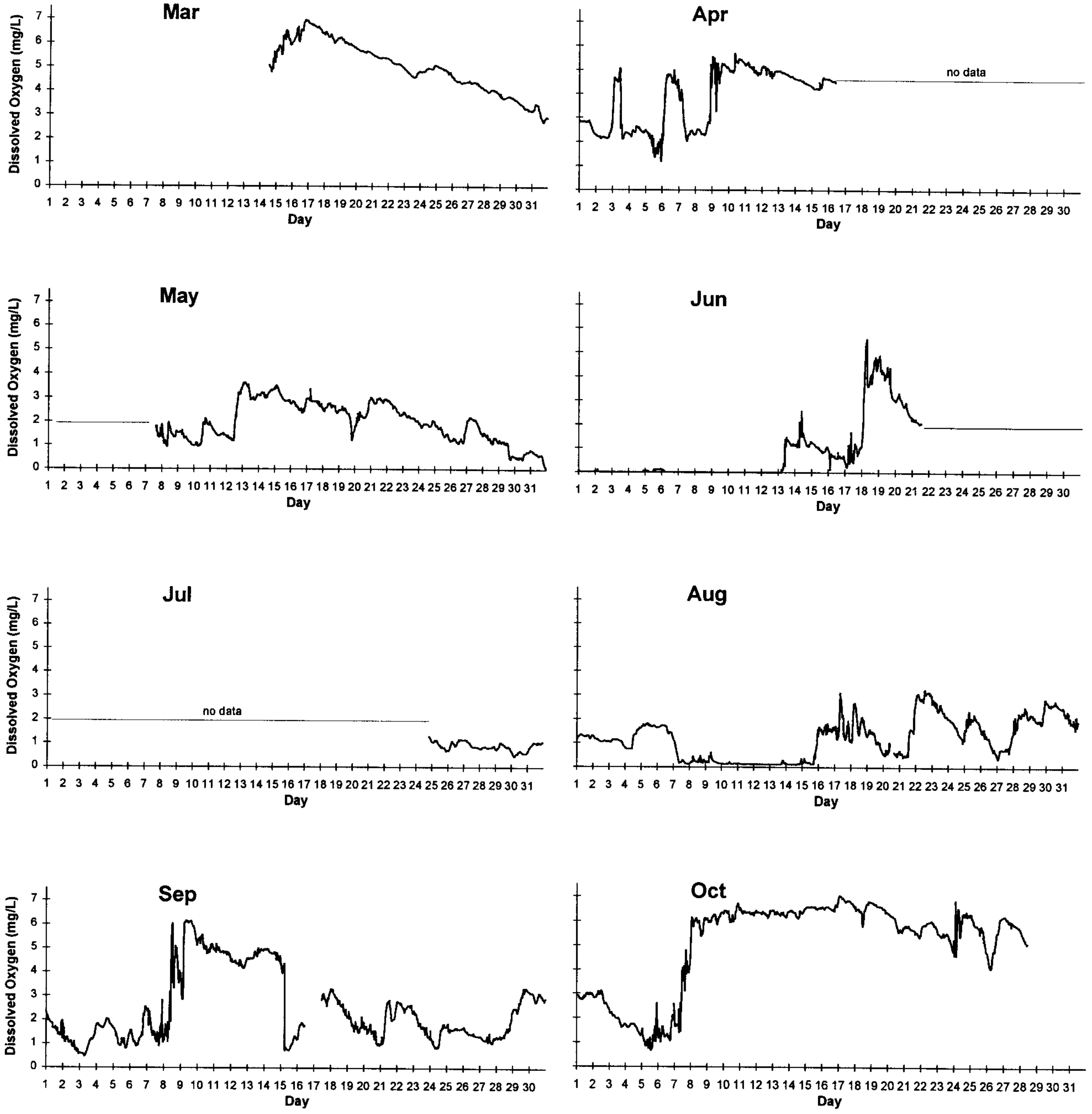

References

Alexander, R.B., P.S. Murdoch, and R.A. Smith. 1996. Streamflow-induced variations in nitrate flux in tributaries to the Atlantic coastal zone. *Biogeochemistry* 33:149–77.

Ammerman, J.W. 1992. Seasonal variation in phosphate turnover in the Mississippi River plume and the inner Gulf shelf: Rapid summer turnover. In *Proceedings: Nutrient Enhanced Coastal Ocean Productivity Workshop*, 69–75. Publication no. TAMU-SG-92-109. College Station, TX: Texas A&M University, Texas Sea Grant College Program.

Ammerman, J.W., W.B. Glover, R.H.S. Ruvalcaba, and J.J.D. MacRae. 1995. Continuous underway measurement of microbial enzyme activities in surface waters of the Mississippi River plume and the Louisiana shelf. In *Proceedings of 1994 Synthesis Workshop, Nutrient-Enhanced Coastal Ocean Productivity Program*, 1–8. Baton Rouge LA: Louisiana State University, Louisiana Sea Grant College Program.

Andersson, L., and L. Rydberg. 1987. Trends in nutrient and oxygen conditions within the Kattegat: Effects of local nutrient supply. *Estuarine, Coastal and Shelf Science* 26:559–79.

Atkinson, L.P., and D. Wallace. 1975. The source of unusually low surface salinities in the Gulf Stream off Georgia. *Deep-Sea Research* 23:913–16.

Atwood, D.K., A. Bratkovich, M. Gallagher, and G.L. Hitchcock. 1994. Introduction to the dedicated issue. *Estuaries* 17:729–31.

Bedinger, C.A., Jr., R.E. Childers, J. Cooper, K.T. Kimball, and A. Kwok. 1981. *Background, program organization and study plan. Ecological investigations of petroleum production platforms in the central Gulf of Mexico*, vol. 1. Report to Bureau of Land Management, contract no. AA551-CT-8-17. San Antonio, TX: Southwest Research Institute.

Bien, G.S., D.E. Contois, and W.H. Thomas. 1958. The removal of soluble silica from fresh water entering the sea. *Geochimica et Cosmochimica Acta* 14:35–54.

Bienfang, P.K., P.J. Harrison, and L.M. Quarmby. 1982. Sinking rate response to depletion of nitrate, phosphate and silicate in four marine diatoms. *Marine Biology* 67:295–302.

Bierman, V.J., Jr., S.C. Hinz, D. Zhu, W.J. Wiseman, Jr., N.N. Rabalais, and R.E. Turner. 1994. A preliminary mass balance model of primary productivity and dissolved oxygen in the Mississippi River plume/Inner Gulf shelf region. *Estuaries* 17:886–99.

Blackwelder, P., T. Hood, C. Alvarez–Zarikian, T.A. Nelsen, and B. McKee. 1996. Benthic foramini- fera from the NECOP study area impacted by the Mississippi River plume and seasonal hypoxia. *Quaternary International* 31:19–36.

Bode, A., and Q. Dortch. 1996. Uptake and regeneration of inorganic nitrogen in coastal waters influenced by the Mississippi River: Spatial and seasonal variations. *Journal of Plankton Research* 18:2251–68.

Boesch, D.F., and N.N. Rabalais. 1991. Effects of hypoxia on continental shelf benthos: Comparisons between the New York Bight and the northern Gulf of Mexico. In *Modern and ancient continental shelf anoxia*, ed. R.V. Tyson and T.H. Pearson, 27–34. Geological Society Special Publication no. 58. London, England: The Geological Society.

Boesch, D.F., R.B. Brinsfield, and R.E. Magnien. In press. Chesapeake Bay eutrophication: Science understanding, ecosystem restoration and challenges for agriculture. *Journal of Environmental Quality.*

Boyles, B., Jr., and Q. Humphries. 1994. Lower Mississippi River streamflow. In *Coastal oceanographic effects of summer 1993 Mississippi River flooding*, ed. M.J. Dowgiallo, 28–29. Special National Oceanic and Atmospheric Administration Report. Silver Spring, MD: NOAA, Coastal Ocean Office/National Weather Service.

Boynton, W.R., J.H. Garber, R. Summers, and W.M. Kemp. 1995. Inputs, transformations, and transport of nitrogen and phosphorus in Chesapeake Bay and selected tributaries. *Estuaries* 18:285–314.

Bratkovich, A, S.P. Dinnel, and D.A. Goolsby. 1994. Variability and prediction of freshwater and nitrate fluxes for the Louisiana–Texas shelf: Mississippi and Atchafalaya River source functions. *Estuaries* 17:766–78.

Brongersma–Sanders, M. 1957. Mass mortality in the sea. In *Treatise on marine ecology and paleoecology*, vol. 1, ed. J.W. Hedgpeth, 941–1010. Geological Society of America, Memoir 67. Baltimore, MD: Waverly Press.

Bryan, C.F., D.A. Rutherford, and B. Walker–Bryan. 1992. Acidification of the lower Mississippi River. *Transactions of the American Fisheries Society* 121:369–77.

Butler, M., and H.G. Dam. 1994. Production rates and characteristics of fecal pellets of the copepod *Acartia tonsa* under simulated phytoplankton bloom conditions: Implications for vertical fluxes. *Marine Ecology Progress Series* 114:81–91.

Carpenter, S.R., N.F. Caraco, D.L. Correll, R.W. Howarth, A.N. Sharpley, and V. H. Smith. 1998. Nonpoint pollution of surface waters with phosphorus and nitrogen. *Ecological Applications* 8:559–68.

Cherfas, J. 1990. The fringe of the ocean: Under siege from land. *Nature* 248:163–65.

Cochrane, J.D., and F.J. Kelly. 1986. Low-frequency circulation on the Texas–Louisiana continental shelf. *Journal of Geophysical Research* 91:10,645–59.

Conley, D.J., S.S. Kolham, and E. Theriot. 1989. Differences in silica content between marine and freshwater diatoms. *Limnology and Oceanography* 34:205–13.

Conley, D.J., C.L. Schelske, and E. F. Stoermer. 1993. Modification of the biogeochemical cycle of silica with eutrophication. *Marine Ecology Progress Series* 101:179–92.

Conseil Permanent International pour l'Exploration de la Mer. 1936. *Bulletin Hydrographique pour l'Année 1935*, Series B1. Charlottelund Slot, Denmark: Le Bureau du Conseil, Service Hydrographique.

Cooper, S.R. 1995. Chesapeake Bay watershed historical land use: Impact on water quality and diatom communities. *Ecological Applications* 5:703–23.

Cooper, S.R., and G.S. Brush. 1991. Long-term history of Chesapeake Bay anoxia. *Science* 254:992–96.

Cooper, S.R., and G.S. Brush. 1993. A 2500 year history of anoxia and eutrophication in Chesapeake Bay. *Estuaries* 16:617–26.

Corner, E.D., R.N. Head, and C.C. Kilvington. 1972. On the nutrition and metabolism of zooplankton. No. 8 of the series The grazing of *Biddulphia* cells by *Calanus helgolandicus*. *Journal of the Marine Biological Association, United Kingdom* 52:847–61.

Coutant, C.C., and D.L. Benson. 1990. Summer habitat suitability for striped bass in Chesapeake Bay: Reflections on a population decline. *Transactions of the American Fisheries Society* 119:757–78.

Cowan, J.L.W., and W.R. Boynton. 1996. Sediment-water oxygen and nutrient exchanges along the longitudinal axis of Chesapeake Bay: Seasonal patterns, controlling factors and ecological significance. *Estuaries* 19:562–80.

Crout, R.L., Wm. J. Wiseman, Jr., and W.-S. Chuang. 1984. Variability of wind-driven currents, west Louisiana inner continental shelf: 1978–1979. *Contributions in Marine Science* 27:1–11.

D'Elia, C.J., J.G. Sanders, and W.R. Boynton. 1986. Nutrient enrichment studies in a coastal plain estuary: Phytoplankton growth in large-scale, continuous cultures. *Canadian Journal of Aquatic and Fisheries Sciences* 43:397–406.

Diaz, R.J., and R. Rosenberg. 1995. Marine benthic hypoxia: A review of its ecological effects and the behavioural responses of benthic macrofauna. *Oceanography and Marine Biology: An Annual Review* 33:245–303.

Diaz, R.J., and A. Solow. 1999. *Ecological and economic consequences of hypoxia: Topic 2 report for the integrated assessment on hypoxia in the Gulf of* Mexico. NOAA Coastal Ocean Program Decision Analysis Series No. 16. Silver Spring, MD: National Oceanic and Atmospheric Administration, Coastal Ocean Office.

Dickey, R.W., GA Fryxell, H.R. Granade, and D. Roelke. 1992. Detection of the marine toxins okadaic acid and domoic acid in shellfish and phytoplankton in the Gulf of Mexico. *Toxicon* 30:355–59.

DiMego, G.J., L.F. Bosart, and G.W. Endersen. 1976. An examination of the frequency and mean conditions surrounding frontal incursions into the Gulf of Mexico and Caribbean Sea. *Monthly Weather Review* 104:709–18.

Dinnel, S.P., and W.J. Wiseman, Jr. 1986. Fresh-water on the Louisiana and Texas shelf. *Continental Shelf Research* 6:765–84.

Dortch, Q. 1994. Changes in phytoplankton numbers and species composition. In *Coastal oceanographic effects of summer 1993 Mississippi River flooding*, ed. M.J. Dowgiallo, 46–49. Special National Oceanic and Atmospheric Administration Report. Silver Spring, MD: NOAA Coastal Ocean Office/National Weather Service.

Dortch, Q. 1998. Phytoplankton characteristics. In *An observational study of the Mississippi–Atchafalaya coastal plume. Final report*, ed. S.P. Murray, 239–68. OCS Study MMS 98-0040. New Orleans, LA: U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region.

Dortch, Q., and T.E. Whitledge. 1992. Does nitrogen or silicon limit phytoplankton production in the Mississippi River plume and nearby regions? *Continental Shelf Research* 12:1293–1309.

Dortch, Q., A. Bode, and R.R. Twilley. 1992a. Nitrogen uptake and regeneration in surface waters of the Louisiana continental shelf influenced by the Mississippi River. In *Proceedings,* *Nutrient Enhanced Coastal Ocean Productivity Workshop*, 52–56. Publ. No. TAMU-SG-92-109. College Station, TX: Texas A&M University, Texas Sea Grant College Program.

Dortch, Q., D. Milsted, N.N. Rabalais, S.E. Lohrenz, D.G. Redalje, M.J. Dagg, R.E. Turner, and T.E. Whitledge. 1992b. Role of silicate availability in phytoplankton species composition and the fate of carbon. In *Proceedings, Nutrient Enhanced Coastal Ocean Productivity Workshop*, 76–83. Publ. no. TAMU-SG-92-109. College Station, TX: Texas A&M University, Texas Sea Grant College Program.

Dortch, Q., N.N. Rabalais, R.E. Turner, and G.T. Rowe. 1994. Respiration rates and hypoxia on the Louisiana shelf. *Estuaries* 17:862–72.

Dortch, Q., D.M. Nelson, R.E. Turner, and N.N. Rabalais. 1995. Silicate limitation on the Louisiana continental shelf. In *Proceedings of 1994 synthesis workshop, nutrient-enhanced coastal ocean productivity program*, 34–39. Baton Rouge, LA: Louisiana State University, Louisiana Sea Grant College Program.

Dortch, Q., N.N. Rabalais, R.E. Turner, and D.M. Nelson. 1997a. Louisiana shelf: Silicon availability, phytoplankton species composition, vertical flux and hypoxia. Invited abstract. ASLO annual meeting, February 10–14. Santa Fe, NM.

Dortch, Q., R. Robichaux, S. Pool, D. Milsted, G. Mire, N.N. Rabalais, T.M. Soniat, G.A. Fryxell, R.E. Turner, and M.L. Parsons. 1997b. Abundance and vertical flux of *Pseudo-nitzschia* in the northern Gulf of Mexico. *Marine Ecology Progress Series* 146:249–64.

Dortch, Q., N.N. Rabalais, R.E. Turner, and N.A. Qureshi. 1998. Invited talk, Hypoxia Effects Workshop, March 11–13, Baton Rouge, Louisiana.

Dortch, Q., M.L. Parsons, N.N. Rabalais, and R.E. Turner. Accepted. What is the threat of harmful algal blooms in Louisiana coastal waters? In Proceedings of recent research in coastal Louisiana: Natural system function and response to human influences, February 3–5, 1998, Lafayette, LA.

Dortch, Q., C.A. Moncreiff, W. Mendenhall, M.L. Parsons, J.S. Franks, and K.W. Hemphill. In press. Spread of *Gymnodinium breve* into the northern Gulf of Mexico. Proceedings of VIII International Conference on Harmful Algae, June 25–29, 1997, Vigo, Spain.

Dortch, Q., J. Rodney, B. Barker, and T. Peterson. In preparation. Ammonium and silicate analyses in turbid waters of variable salinity. *Marine Chemistry*.

Dortch, Q., P. Ward, and R. Robichaux. Unpublished data. Bioassays using *in vivo* fluorescence and phytoplankton cell counts to determine nutrient limitation in the plume of the Mississippi River.

Dowgiallo, M.J., ed. 1994. *Coastal oceanographic effects of summer 1993 Mississippi River flooding.* Special National Oceanic and Atmospheric Administration Report. Silver Spring, MD: NOAA Coastal Ocean Office/National Weather Service.

Dunn, D.D. 1996. *Trends in nutrient inflows to the Gulf of Mexico from streams draining the conterminous United States 1972–1993*. U.S. Geological Survey, Water-Resources Investigations Report 96—4113. Prepared in cooperation with the U.S. Environmental Protection Agency, Gulf of Mexico Program, Nutrient Enrichment Issue Committee. Austin, TX: U.S. Department of the Interior, U.S. Geological Survey.

Eadie, B.J., B.A. McKee, M.B. Lansing, J.A. Robbins, S. Metz, and J.H. Trefry. 1994. Records of nutrient-enhanced coastal productivity in sediments from the Louisiana continental shelf. *Estuaries* 17:754–65.

Eleuterius, L., H. Perry, C. Eleuterius, J. Warren, and J. Caldwell. 1981. Causative analysis on a nearshore bloom of *Oscillatoria erythraea (Trichodesmium)* in the northern Gulf of Mexico. *Northeast Gulf Science* 5:1–11.

Everett, D. 1971. Hydrologic and water quality characteristics of the lower Mississippi River. Louisiana Department of Public Works, Technical Report no. 5. Baton Rouge, LA.

Faganeli, J., A. Avcin, N. Fanuko, A. Malej, V. Turk, P. Tusnik, B. Vriser, and A. Vukovic. 1985. Bottom layer anoxia in the central part of the Gulf of Trieste in the late summer of 1983. *Marine Pollution Bulletin* 16:75–78.

Fahnenstiel, G.L., M.J. McCormick, G.A. Lang, D.G. Redalje, S.E. Lohrenz, M. Markowitz, B. Wagoner, and H.J. Carrick. 1995. Taxon-specific growth and loss rates for dominant phytoplankton populations from the northern Gulf of Mexico. *Marine Ecology Progress Series* 117:229–39.

Fisher, T.R., L.W. Harding, Jr., D.W. Stanley, and L.G. Ward. 1988. Phytoplankton, nutrients, and turbidity in the Chesapeake, Delaware, and Hudson estuaries. *Estuarine and Coastal Shelf Science* 27:61–93.

Fisher, T.R., J.M. Melack, J.U. Brobbelaar, and R.W. Howarth. 1995. Nutrient limitation of phytoplankton and eutrophication of inland, estuarine, and marine waters. In *Phosphorus in the global environment*, ed. H. Tiessen, 301–22. Chichester, England: John Wiley & Sons, Ltd.

Fox, L.E., S.L. Sager, and S.C. Wofsy. 1985. Factors controlling the concentrations of soluble phosphorus in the Mississippi estuary. *Limnology and Oceanography* 30:826–32.

Fransz, H.G., and J.H.G. Verhagen. 1985. Modeling research on the production cycle of phytoplankton in the southern bight of the North Sea in relation to riverborne nutrient loads. *Netherlands Journal of Sea Research* 19: 241–50.

Fry, B., R.K. Anderson, L. Entzeroth, J.L. Bird, and P.L. Parker. 1984. ^{13}C enrichment and oceanic food web structure in the northwestern Gulf of Mexico. *Contributions in Marine Science* 27:13–47.

Fu, J.-M., and J.W. Winchester. 1994. Sources of nitrogen in three watersheds of northern Florida, USA: Mainly atmospheric deposition. *Geochimica et Cosmochimica Acta* 58:1581–90.

Fucik, K.W. 1974. The effect of petroleum operations on the phytoplankton ecology of the Louisiana coastal waters. Master’s thesis, Texas A&M University, College Station, TX.

Gardner, W.S., R. Benner, G. Chin-Leo, J.B. Cotner, Jr., B.J. Eadie, J.F. Cavaletto, and M.B. Lansing. 1994. Mineralization of organic material and bacterial dynamics in Mississippi River plume water. *Estuaries* 17:816–28.

Gardner, W.S., J.F. Cavaletto, J.B. Cotner, and J.M. Johnson. 1997. Effects of natural light on nitrogen cycling rates in the Mississippi River plume. *Limnology and Oceanography* 42:273–81.

Garside, C., and T.C. Malone. 1978. Monthly oxygen and carbon budgets of the New York Bight apex. *Estuarine, Coastal and Shelf Science* 6:93–104.

Gaston, G.R. 1985. Effects of hypoxia on macrobenthos of the inner shelf off Cameron, Louisiana. *Estuarine, Coastal and Shelf Science* 20:603–13.

Gaston, G.R., P.A. Rutledge, and M.L. Walther. 1985. The effects of hypoxia and brine on recolonization by macrobenthos off Cameron, Louisiana (USA). *Contributions in Marine Science* 28:29–93.

Goldman, J.C., and P.M. Glibert. 1983. Kinetics of inorganic nitrogen uptake by phytoplankton. In *Nitrogen in marine environment*, ed. E.J. Carpenter and D.G. Capone, 223–74. New York, NY: Academic Press.

Goolsby, D.A. 1994. Flux of herbicides and nitrate from the Mississippi River to the Gulf of Mexico. In *Coastal oceanographic effects of summer 1993 Mississippi River flooding*, ed. M.J. Dowgiallo, 32–35. Special National Oceanic and Atmospheric Administration Report. Silver Spring, MD: NOAA Coastal Ocean Office/National Weather Service.

Goolsby, D.A., W.A. Battaglin, and E.M. Thurman. 1993. *Occurrence and transport of agriculture chemicals in the Mississippi River Basin, July through August 1993*. U.S. Geological Survey Circular 1120-C. Denver, CO: U.S. Department of the Interior, U.S. Geological Survey.

Goolsby, D.A., W.A. Battaglin, G.B. Lawrence, R.S. Artz, B.T. Aulenbach, and R.P. Hooper. 1999. *Flux and sources of nutrients in the Mississippi–Atchafalaya River Basin: Topic 3 report for the integrated assessment on hypoxia in the Gulf of Mexico.* NOAA Coastal Ocean Program Decision Analysis Series No. 17. Silver Spring, MD: NOAA Coastal Ocean Office.

Gulf States Marine Fisheries Commission. 1982 et seq. *SEAMAP environmental and biological atlas of the Gulf of Mexico, 1982*. Ocean Springs, MS: Gulf States Marine Fisheries Commission.

Gutierrez de Velasco, G., and C.D. Winant. 1996. Seasonal patterns of wind stress and wind stress curl over the Gulf of Mexico. *Journal of Geophysical Research* 101(C8):18,127–140.

Hallegraeff, G.M. 1993. A review of harmful algal blooms and their apparent global increase. *Phycologia* 32:79–99.

Harding, L.W., Jr., and E.S. Perry. 1997. Long-term increase of phytoplankton biomass in Chesapeake Bay, 1950–1994. *Marine Ecology Progress Series* 157:39–52.

Harper, D.E., Jr., R.R. Salzer, and R.J. Case. 1981. The occurrence of hypoxic bottom water off the upper Texas coast and its effect on the benthic biota. *Contributions in Marine Science* 24:53–79.

Harper, D.E., Jr., L.D. McKinney, J.M. Nance, and R.R. Salzer. 1991. Recovery responses of two benthic assemblages following an acute hypoxic event on the Texas continental shelf, northwestern Gulf of Mexico. In *Modern and ancient continental shelf anoxia*, ed. R.V. Tyson and T.H. Pearson, 49–64. Geological Society Special Publication no. 58. London, England: The Geological Society.

Harris, G.P. 1986. *Plankton ecology: Structure, function and fluctuations*. New York, NY: Chapman and Hall.

Harrison, P.J., H.L. Conway, R.W. Holmes, and C.O. Davis. 1977. Marine diatoms in chemostats under silicate or ammonium limitation. III: Cellular chemical composition and morphology of three diatoms. *Marine Biology* 43:19–31.

Hendee, J.C. 1994. Data management for the nutrient enhanced coastal ocean productivity program. *Estuaries* 17:900–3.

Hirsch, R.M., J.R. Slack, and R.A. Smith. 1982. Techniques of trend analysis for monthly water quality data. *Water Resources Research* 18:107–21.

Hitchcock, G.L., W.J. Wiseman, Jr., W.C. Boicourt, A.J. Mariano, N. Walker, T.A. Nelsen, and E. Ryan. 1997. Property fields in an effluent plume of the Mississippi River. *Journal of Marine Systems* 12:109–26.

Howarth, R.W., G. Billen, D. Swaney, A. Townsend, N. Jaworski, K. Lajtha, J.A. Downing, R. Elmgren, N. Caraco, T. Jordan, F. Berendse, J. Freney, V. Kudeyarov, P. Murdoch, and Z. Zhao-Liang. 1996. Regional nitrogen budgets and riverine N & P fluxes for the drainages to the North Atlantic Ocean: Natural and human influences. *Biogeochemistry* 35:75–79.

Humborg, C., V. Ittekkot, A. Caciasu, and B. v. Bodungen. 1997. Effect of Danube River dam on Black Sea biogeochemistry and ecosystem structure. *Nature* 386:385–88.

Jackson, W.B., and G.M. Faw, eds. 1980. Biological/chemical survey of Texoma and Capline sector salt dome brine disposal sites off Louisiana, 1978–1979. NOAA Techn. Memo NMFS-SEFC-29, NOAA/NMFS Final Report to DOE. Galveston, TX: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Johansson, J.O.R., and R.R. Lewis III. 1992. Recent improvements in water quality and biological indicators in Hillsborough Bay, a highly impacted subdivision of Tampa Bay, Florida, USA. In *Marine coastal eutrophication: The response of marine transitional systems to human impact Problems and perspectives for restoration*, ed. R.W. Vollenweider, R. Marchetti, and R. Viviani, 1,191–1,215. New York, NY: Elsevier Science Publishing Co.

Justić, D. 1991a. Hypoxic conditions in the northern Adriatic Sea: Historical development and ecological significance. In *Modern and ancient continental shelf anoxia*, ed. R.V. Tyson and T.H. Pearson, 95–105. Geological Society Special Publication no. 58. London, England: The Geological Society.

Justić, D. 1991b. A simple oxygen index for trophic state description. *Marine Pollution Bulletin* 22:201–4.

Justić, D., T. Legovic, and L. Rottini–Sandrini. 1987. Trends in oxygen content, 1911–84, and occurrence of benthic mortality in the northern Adriatic Sea. *Estuarine and Coastal Shelf Science* 24:435–45.

Justić, D., N.N. Rabalais, R.E. Turner, and W.J. Wiseman, Jr. 1993. Seasonal coupling between riverborne nutrients, net productivity and hypoxia. *Marine Pollution Bulletin* 26:184–89.

Justić, D., N.N. Rabalais, and R.E. Turner. 1994. Riverborne nutrients, hypoxia and coastal ecosystem evolution: Biological responses to long-term changes in nutrient loads carried by the Po and the Mississippi Rivers. In *Changes in fluxes in estuaries: Implications from science to management*, ed. K.R. Dyer and R.J. Orth, 161–67. Proceedings of ECSA22/ERF Symposium, International Symposium Series. Fredensborg, Denmark: Olsen & Olsen.

Justić, D., N.N. Rabalais, and R.E. Turner. 1995a. Stoichiometric nutrient balance and origin of coastal eutrophication. *Marine Pollution Bulletin* 30:41–46.

Justić, D., N.N. Rabalais, R.E. Turner, and Q. Dortch. 1995b. Changes in nutrient structure of river-dominated coastal waters: Stoichiometric nutrient balance and its consequences. *Estuarine, Coastal and Shelf Science* 40:339–56.

Justić, D., N.N. Rabalais, and R.E. Turner. 1996. Effects of climate change on hypoxia in coastal waters: A doubled CO_2 scenario for the northern Gulf of Mexico. *Limnology and Oceanography* 41:992–1003.

Justić, D., N.N. Rabalais, and R.E. Turner. 1997. Impacts of climate change on net productivity of coastal waters: Implications for carbon budget and hypoxia. *Climate Research* 8:225–37.

Kelly, F.J., J.E. Schmitz, R.E. Randall, and J.D. Cochrane. 1983. Physical oceanography. In *Evaluation of brine disposal from the Bryan Mound site of the Strategic Petroleum Reserve Program: Annual report for September 1981 through August 1982*, vol. 1, ed. R.W. Hann, Jr., and R.E. Randall. U.S. Department of Energy, Contract No. DE-FC96-79P010114. College Station, TX: Texas A&M University.

Kelly, F.J., J.E. Schmitz, and R.E. Randall. 1984. Physical oceanography. In *Offshore oceanographic and environmental monitoring service for the Strategic Petroleum Reserve: Eighteen-month report for the West Hackberry site from May 1982 through November 1983*, ed. R.W. Hann, Jr., C.P. Giammona, and R.E. Randall. U.S. Department of Energy, contract no. DO-P010850-3. Lake Charles, LA: McNeese State University.

Keown, M.P., E.A. Dardeau, Jr., and E.M. Causey. 1986. Historic trends in the sediment flow regime of the Mississippi River. *Water Resources Research* 22:1555–64.

Kesel, R.H. 1988. The decline in the suspended load of the lower Mississippi River and its influence on adjacent wetlands. *Environmental Geology and Water Science* 11:271–81.

Kimsey, J.B., and R.F. Temple. 1963. Currents on the continental shelf of the northwestern Gulf of Mexico. *U.S. Bureau of Commercial Fisheries Circular* 161:23–27.

Kimsey, J.B., and R.F. Temple. 1964. Currents on the continental shelf of the northwestern Gulf of Mexico. *U.S. Bureau of Commercial Fisheries Circular* 183:25–27.

Larsson, U.R., R. Elmgren, and F. Wulff. 1985. Eutrophication and the Baltic Sea: Causes and consequences. *Ambio* 14:9–14.

Leenheer, J. 1982. United States Geological Survey Information Service. In *Transport of carbon and minerals in major world rivers*, part 1, ed. E.T. Degens. Mitt. Geol.-Paläont. Inst. Univ. Hamburg, SCOPE/UNEP Sonderbd. 52:355–56.

Leming, T.D., and W.E. Stuntz. 1984. Zones of coastal hypoxia revealed by satellite scanning have implications for strategic fishing. *Nature* 310:136–38.

Limno-Tech, Inc. 1995. *Estimated responses of water quality on the Louisiana inner shelf to nutrient load reductions in the Mississippi and Atchafalaya Rivers*. Revised Final Report to U.S. Environmental Protection Agency, Gulf of Mexico Program Office. John C. Stennis Space Center, MS.

Loder, T.C., and R.P. Reichard. 1981. The dynamics of conservative mixing in estuaries. *Estuaries* 4:64–69.

Loesch, H. 1960. Sporadic mass shoreward migrations of demersal fish and crustaceans in Mobile Bay, Alabama. *Ecology* 41:292–98.

Lohrenz, S.E., M.J. Dagg, and T.E. Whitledge. 1990. Enhanced primary production at the plume/ oceanic interface of the Mississippi River. *Continental Shelf Research* 10:639–64.

Lohrenz, S.E., G.L. Fahnenstiel, and D.G. Redalje. 1994. Spatial and temporal variations in photosynthesis parameters in relation to environmental conditions in coastal waters of the northern Gulf of Mexico. *Estuaries* 17:779–95.

Lohrenz, S.E., G.L. Fahnenstiel, D.G. Redalje, G.A. Lang, X. Chen, and M.J. Dagg. 1997. Variations in primary production of northern Gulf of Mexico continental shelf waters linked to nutrient inputs from the Mississippi River. *Marine Ecology Progress Series* 155:435–54.

Lohrenz, S.E., G.L. Fahnenstiel, D.G. Redalje, G.A. Lang, M J. Dagg, T.E. Whitledge, and Q. Dortch. 1999. The interplay of nutrients, irradiance and mixing as factors regulating primary production in coastal waters impacted by the Mississippi River plume. *Continental Shelf Research* 19:1113–41.

Lopez–Veneroni, D., and L.A. Cifuentes. 1994. Transport of dissolved organic nitrogen in Mississippi River plume and Texas–Louisiana continental shelf near-surface waters. *Estuaries* 17:796–808.

Makulla, A., and U. Sommer. 1993. Relationships between resource ratios and phytoplankton species composition during spring in five north German lakes. *Limnology and Oceanography* 38:846–56.

Malone, T.C. 1991. River flow, phytoplankton production and oxygen depletion in Chesapeake Bay. In *Modern and ancient continental shelf anoxia*, ed. R.V. Tyson and T.H. Pearson, 83–93. Geological Society Special Publication no. 58. London, England: The Geological Society.

Malone, T.C. 1992. Effects of water column processes on dissolved oxygen, nutrients, phytoplankton and zooplankton. In *Oxygen dynamics in Chesapeake Bay: A synthesis of recent research*, ed. D.E. Smith, M. Leffler, and G. Makiernan, 61–112. College Park, MD: Maryland Sea Grant Program.

Malone, T.C., W. Boynton, T. Horton, and C. Stevenson. 1993. Nutrient loading to surface waters: Chesapeake case study. In *Keeping pace with science and engineering*, ed. M.F. Uman, 8–38. Washington, DC: National Academy Press.

May, E.B. 1973. Extensive oxygen depletion in Mobile Bay, Alabama. *Limnology and Oceanography* 18:353–66.

Mayer, L.M., R.G. Keil, S.A. Macko, S.B. Joye, K.C. Ruttenberg, and R.C. Aller. 1998. Importance of suspended particulates in riverine delivery of bioavailable nitrogen to coastal zones. *Global Biogeochemical Cycles* 12(4):573–79.

McNaughton, A. 1998. The evolution of inshore hydrography of the West Louisiana inner shelf, 1992. Master's thesis, Department of Oceanography and Coastal Sciences, Louisiana State University, Baton Rouge, LA.

Meade, R.H., ed. 1995. *Contaminants in the Mississippi River, 1987–1992*. U.S. Geological Survey Circular 1133. Denver, CO: U.S. Department of the Interior, U.S. Geological Survey.

Meade, R.H., and R. Parker. 1985. Sediment in rivers of the United States. In *National water summary 1984, 49–60*. U.S. Geological Survey Water Supply Paper no. 2275. Denver, CO: U.S. Department of the Interior, U.S. Geological Survey.

Meade, R.H., T. Yuzyk, and T. Day. 1990. Movement and storage of sediment in rivers of the United States and Canada. In *Surface water hydrology*, ed. M.G. Wolman and H.C. Riggs, 255–80, vol. 0-1 of *The geology of North America*, ed. W.H. Riggs. Boulder, CO: Geological Society of America.

Meybeck, M. 1982. Carbon, nitrogen and phosphorus transport by world rivers. *American Journal of Science* 282:401–50.

Miller, J.R., and G.L. Russell. 1992. The impact of global warming on river runoff. *Journal of Geophysical Research* 97:2757–64.

Millie, D.F., C.P. Dionigi, and C.M. Hersh. 1994. Potential effects of triazine herbicides on marine phytoplankton. In *Coastal oceanographic effects of summer 1993 Mississippi River flooding*, ed. M.J. Dowgiallo, 76–77. Special National Oceanic and Atmospheric Administration Report. Silver Spring, MD: NOAA Coastal Ocean Office/National Weather Service.

Milliman, J.D., and R.H. Meade. 1983. World-wide delivery of river sediment to the ocean. *Journal of Geology* 91:1–21.

Moore, W.S. 1996. Large groundwater inputs to coastal waters revealed by 226 Ra enrichments. *Nature* 380:612–14.

Moshiri, G.A., N.G. Auman, and W.B. Crumpton. 1981. Reversal of the eutrophication process: A case study. In *Estuaries and nutrients*, ed. B.J. Neilson and L.E. Cronin, 373–90. Clifton, NJ: Humana Press, Inc.

Murray, S.P., ed. 1998. *An observational study of the Mississippi–Atchafalaya coastal plume. Final report.* OCS Study MMS 98-0040. New Orleans, LA: U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region.

Murray, S.P., E. Jarosz, and E.T. Weeks III. 1998. Physical oceanographic observations of the coastal plume. in *An observational study of the Mississippi–Atchafalaya coastal plume. Final report*, ed. S.P. Murray, 5–105. OCS Study MMS 98-0040. New Orleans, LA: U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region.

Nelsen, T.A., P. Blackwelder, T. Hood, B. McKee, N. Romer, C. Alvarez–Zarikian, and S. Metz. 1994. Time-based correlation of biogenic, lithogenic and authigenic sediment components with anthropogenic inputs in the Gulf of Mexico NECOP study area. *Estuaries* 17:873–85.

Nelson, D.M., and M.A. Brzezinski. 1990. Kinetics of silicic acid uptake by natural diatom assemblages in two Gulf Stream warm-core rings. *Marine Ecology Progress Series* 62:283–92.

Nelson, D.M., and Q. Dortch. 1996. Silicic acid depletion and silicon limitation in the plume of the Mississippi River: Evidence from kinetic studies in spring and summer. *Marine Ecology Progress Series* 136:163–78.

Officer, C.B., and J.H. Ryther. 1980. The possible importance of silicon in marine eutrophication. *Marine Ecology Progress Series* 3:83–91.

Officer, C.B., R.B. Biggs, J.L. Taft, L.E. Cronin, M.A. Tyler, and W.R. Boynton. 1984. Chesapeake Bay anoxia: Origin, development and significance. *Science* 223:22–27.

Paerl, H., J. Pinckney, J. Fear, and B. Peierls. 1998. Ecosystem responses to internal and watershed organic matter loading: Consequences for hypoxia in the eutrophying Neuse River estuary, NC, USA. *Marine Ecology Progress Series* 166:17–25.

Pakulski, J.D., R. Benner, R. Amon, B. Eadie, and T. Whitledge. 1995. Community metabolism and nutrient cycling in the Mississippi River plume: Evidence for intense nitrification at intermediate salinities. *Marine Ecology Progress Series* 117:207–18.

Parker, C.A., and J.E. O'Reilly. 1991. Oxygen depletion in Long Island Sound: A historical perspective. *Estuaries* 14:248–64.

Parsons, M.L., Q. Dortch, and G.A. Fryxell. In press. A multi-year study of the presence of potential domoic acid-producing *Pseudo-nitzschia* species in the coastal and estuarine waters of Louisiana, USA. Proceedings of VIII International Conference on Harmful Algae, June 25–29, 1997, Vigo, Spain.

Pavela, J.S., J.L. Ross, and M.E. Chittenden. 1983. Sharp reductions in abundance of fishes and benthic macroinvertebrates in the Gulf of Mexico off Texas associated with hypoxia. *Northeast Gulf Science* 6:167–73.

Pokryfki, L., and R.E. Randall. 1987. Nearshore hypoxia in the bottom water of the northwestern Gulf of Mexico from 1981 to 1984. *Marine Environmental Research* 22:75–90.

Qureshi, N.A. 1995. The role of fecal pellets in the flux of carbon to the sea floor on a river-influenced continental shelf subject to hypoxia. Ph.D. diss., Department of Oceanography and Coastal Sciences, Louisiana State University, Baton Rouge, LA.

Rabalais, N.N. 1992. *An updated summary of status and trends in indicators of nutrient enrichment in the Gulf of Mexico*. Environmental Protection Agency Publ. No. EPA/800-R-92-004, Gulf of Mexico Program, Nutrient Enrichment Subcommittee, John C. Stennis Space Center, MS.

Rabalais, N.N. 1998a (on line). Oxygen depletion in coastal waters. In *NOAA's state of the coast report.* Silver Spring, MD: National Oceanic and Atmospheric Administration. URL: http://state_of_coast. noaa.gov/bulletins/html/hyp_09/hyp.html.

Rabalais, N.N. 1998b. Hypoxia. In *An observational study of the Mississippi–Atchafalaya coastal plume*. *Final report*, ed. S.P. Murray, 325–54. OCS Study MMS 98-0040. New Orleans, LA: U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region.

Rabalais, N.N., and R.E. Turner. 1998. Pigment and nutrient distributions. In *An observational study of the Mississippi–Atchafalaya coastal plume*. *Final report*, ed. S.P. Murray, 208–30. OCS Study MMS 98-0040. New Orleans, LA: U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region.

Rabalais, N.N., R.E. Turner, W.J. Wiseman, Jr., and D.F. Boesch. 1986. *Hydrographic, biological, and nutrient characteristics of the water column on the Louisiana shelf, July and September 1985*. Data Report no. 3. Cocodrie, LA: Louisiana Universities Marine Consortium.

Rabalais, N.N., R.E. Turner, W.J. Wiseman, Jr., and D.F. Boesch. 1991. A brief summary of hypoxia on the northern Gulf of Mexico continental shelf: 1985–1988. In *Modern and ancient continental shelf anoxia*, ed. R.V. Tyson and T.H. Pearson, 35–46. Geological Society Special Publication no. 58. London, England: The Geological Society.

Rabalais, N.N., R.E. Turner, and W.J. Wiseman, Jr. 1992a. Distribution and characteristics of hypoxia on the Louisiana shelf in 1990 and 1991. In *Proceedings, nutrient enhanced coastal ocean productivity workshop*, 15–20. Publication no. TAMU-SG-92-109. College Station, TX: Texas A&M University, Texas Sea Grant College Program.

Rabalais, N.N., R.E. Turner, and Q. Dortch. 1992b. Louisiana continental shelf sediments: Indicators of riverine influence. In *Proceedings; Nutrient enhanced coastal ocean productivity*, 77–81. Publication no. TAMU-SG-92-109. Galveston, TX: Texas Sea Grant College Program.

Rabalais, N.N., W.J. Wiseman, Jr., and R.E. Turner. 1994. Comparison of continuous records of near-bottom dissolved oxygen from the hypoxia zone of Louisiana. *Estuaries* 17:850–61.

Rabalais, N.N., R.E. Turner, D. Justić, Q. Dortch, W.J. Wiseman, Jr., and B.K. Sen Gupta. 1996. Nutrient changes in the Mississippi River and system responses on the adjacent continental shelf. *Estuaries* 19:386–407.

Rabalais, N.N., R.E. Turner, W.J. Wiseman, Jr., and Q. Dortch. 1998. Consequences of the 1993 Mississippi River flood in the Gulf of Mexico. *Regulated Rivers: Research & Management* 14:161–77.

Ragan, J.G., A.H. Harris, and J.H. Green. 1978. Temperature, salinity and oxygen measurements of surface and bottom waters on the continental shelf off Louisiana during portions of 1975 and 1976. *Professional Papers Series (Biology)* 3:1–29. Nicholls State University, Thibodaux, LA.

Redalje, D.G., S.E. Lohrenz, and G.L. Fahnenstiel. 1994. The relationship between primary production and the vertical export of particulate organic matter in a river-impacted coastal ecosystem. *Estuaries* 17:829–38.

Reigman, R. 1992. *Phaeocystis* blooms and eutrophication of the continental coastal zones of the North Sea. *Marine Biology* 112:479–84.

Renaud, M. 1986. Hypoxia in Louisiana coastal waters during 1983: Implications for fisheries. *Fishery Bulletin* 84:19–26.

Rhee, G.Y. 1973. A continuous culture study of phosphate uptake, growth rate and polyphosphate in *Scenedesmus* sp. *Journal of Phycology* 9:495–506.

Richards, F.A. 1957. Oxygen in the ocean. In *Treatise on marine ecology and paleoecology*, ed. J.W. Hedgpeth, 185–238. Geological Society of America Memoir, vol. 67, no. 1. Baltimore, MD: Waverly Press.

Richardson, K. 1996. Conclusion, research and eutrophication control. In *Eutrophication in coastal marine systems*, ed. B.B Jørgensen and K. Richardson, 243–67. Coastal and Estuarine Studies, vol. 52. Washington, DC: American Geophysical Union.

Riisgård, H.U., C. Jürgensen, and F.Ø. Andersen. 1996. Case study: Kertinge Nor. In *Eutrophication in coastal marine systems*, ed. B.B. Jørgensen and K. Richardson, 205–20. Coastal and Estuarine Studies, vol. 52. Washington, DC: American Geophysical Union.

Rosenberg, R. 1972. Benthic faunal recovery in a Swedish fjord following the closure of a sulphite pulp mill. *Oikos* 23:92–108.

Rosenberg, R. 1976. Benthic faunal dynamics during succession following pollution abatement in a Swedish estuary. *Oikos* 27:414–27.

Rosenberg, R., ed. 1986. *A review: Eutrophication in marine waters surrounding Sweden*. National Swedish Environmental Protection Board Report, 3054 (translation of SNV PM 1808 (1984)). Soina, Sweden.

Round, F.E., R.M. Crawford, and D.G. Mann. 1990. *The diatoms: Biology and morphology of the genera*. Cambridge, England: Cambridge University Press.

Schelske, C.L., E.F. Stoermer, D.J. Conley, J.A. Robbins, and R.M. Gloover. 1983. Early eutrophication in the lower Great Lakes: New evidence from biogenic silica in sediments. *Science* 222:320–22.

Schelske, C.L., D.J. Conley, E.F. Stoermer, T.L. Newberry, and C.D. Campbell. 1986. Biogenic silica and phosphorus accumulation in sediments as indices of eutrophication in the Laurentian Great Lakes. *Hydrobiologia* 143:79–86.

Sen Gupta, B.K., and M.L. Machain–Castillo. 1993. Benthic foraminifera in oxygen-poor habitats. *Marine Micropaleontology* 20:183–201.

Sen Gupta, B.K., R.F. Lee, and M.S. May. 1981. Upwelling and an unusual assemblage of benthic foraminifera on the northern Florida continental slope. *Journal of Paleontology* 55:853–57.

Sen Gupta, B.K., R.E. Turner, and N.N. Rabalais. 1996. Seasonal oxygen depletion in continental-shelf waters of Louisiana: Historical record of benthic foraminifers. *Geology* 24:227–30.

Shindler, D.W. 1977. Evolution of phosphorus limitation in lakes. *Science* 195:260–62.

Shindler, D.W. 1978. Factors regulating phytoplankton production and standing crop in the world's freshwaters. *Limnology and Oceanography* 23:322–31.

Shumway, S.E. 1990. A review of the effects of algal blooms on shellfish and aquaculture. *Journal of the World Aquaculture Society* 21:65–104.

Simmons, E.G., and W.H. Thomas. 1962. Phytoplankton of the eastern Mississippi Delta. *Publications of the Institute of Marine Science, University of Texas* 8:269–98.

Simpson, J.H., W.G. Bos, F. Schirmer, A.J. Souza, T.P. Rippeth, S.E. Jones, and D. Hydes. 1993. Periodic stratification in the Rhine ROFI in the North Sea. *Oceanologica Acta* 16:23–32.

Sklar, F.H., and R.E. Turner. 1981. Characteristics of phytoplankton production off Barataria Bay in an area influenced by the Mississippi River. *Contributions in Marine Science* 24:93–106.

Smayda, T.J. 1989. Primary production and the global epidemic of phytoplankton blooms in the sea: A linkage? In *Novel phytoplankton blooms*, ed. E.M. Cosper, V.M. Bricelj, and E.J. Carpenter, 449–83. Berlin, Germany: Springer-Verlag.

Smayda, T.J. 1990. Novel and nuisance phytoplankton blooms in the sea: Evidence for global epidemic. In *Toxic marine phytoplankton*, ed. E. Graneli, B. Sundstrom, R. Edler, and D.M. Anderson, 29–40. New York, NY: Elsevier Science Publishing Co.

Smith, N.P. 1980. On the hydrography of shelf waters off the central Texas Gulf coast. *Journal of Physical Oceanography* 10:806–13.

Smith, R.A., R.B. Alexander, and M.G. Wolman. 1987. Water-quality trends in the nation's rivers. *Science* 235:1605–15.

Smith, S.M., and G.L. Hitchcock. 1994. Nutrient enrichments and phytoplankton growth in the surface waters of the Louisiana Bight. *Estuaries* 17:740–53.

Smith, S.V. 1981. Responses of Kaneohe Bay, Hawaii, to relaxation of sewage stress. In *Estuaries and nutrients*, ed. B.J. Neilson and L.E. Cronin, 391–410. Clifton, NJ: Humana Press, Inc.

Sommer, U. 1989. Nutrient status and nutrient competition of phytoplankton in a shallow, hypertrophic lake. *Limnology and Oceanography* 34:1162–73.

Sommer, U. 1993. Phytoplankton competition in Pluss-see: A field test of the resource-ratio hypothesis. *Limnology and Oceanography* 38:838–45.

Sommer, U. 1994. The impact of light intensity and day length on silicate and nitrate competition among marine phytoplankton. *Limnology and Oceanography* 39:1680–88.

Sommer, U. 1995. Eutrophication related changes in phytoplankton species composition: Is there a role of nutrient competition? ICES-CM-1995/T:7.

Sommer, U. 1996. Nutrient competition experiments with periphyton from the Baltic Sea. *Marine Ecology Progress Series* 140:161–67.

Stuntz, W.E., N. Sanders, T.D. Leming, K.N. Baxter, and R.M. Barazotto. 1982. Area of hypoxic bottom water found in the northern Gulf of Mexico. *Coastal Oceanography and Climatology News* 4(4).

Suess, E. 1980. Particulate organic carbon flux in the oceans: Surface productivity and oxygen utilization. *Nature* 288:260–63.

Swanson, R.L., and C.A. Parker. 1988. Physical environmental factors contributing to recurring hypoxia in the New York Bight. *Transactions of the American Fisheries Society* 117:37–47.

Swanson, R.L., and C.J. Sindermann, eds. 1979*. Oxygen depletion and associated benthic mortalities in New York Bight, 1976*. National Oceanic and Atmospheric Administration Professional Paper 11. Washington, DC.

Telang, S.A., R. Pocklington, A.S. Naidu, E.A. Romankevich, I.I. Gitelson, and M.I. Gladyshev. 1991. Carbon and mineral transport in major North American, Russian, Arctic, and Siberian rivers: The St. Lawrence, the Mackenzie, the Yukon, the Arctic Alaskan rivers, the Arctic basin rivers in the Soviet Union, and the Yenisei. In *Biogeochemistry of major world rivers*, ed. E.T. Degens, S. Kempe, and J.E. Richey, 75–104. SCOPE 42. New York, NY: John Wiley and Sons.

Tester, P.A., and L.P. Atkinson. 1994. Low salinity water in the Gulf Stream off North Carolina. In *Coastal oceanographic effects of summer 1993 Mississippi River flooding*, ed. M.J.

Dowgiallo, 72–76. Special National Oceanic and Atmospheric Administration Report. Silver Spring, MD: NOAA Coastal Ocean Office/National Weather Service.

Thomas, W.H., and E.G. Simmons. 1960. Phytoplankton production in the Mississippi River Delta. In *Recent sediments, northwest Gulf of Mexico*, ed. F.P. Shepard, 103–16. Tulsa, OK: American Association of Petroleum Geologists.

Tilman, D., R.L. Kriesling, R.W. Sterner, S.S. Kilham, and F.A. Johnson. 1986. Green, bluegreen, and diatom algae: Taxonomic differences in competitive availability for phosphorus, silicon, and nitrogen. *Archiv für Hydrobiologie* 106:473–85.

Tolmazin, R. 1985. Changing coastal oceanography of the Black Sea. I: Northwestern shelf. *Progress in Oceanography* 15:217–76.

Townsend, D.W., and L.M. Cammen. 1988. Potential importance of the timing of spring plankton blooms to benthic-pelagic coupling and recruitment of juvenile demersal fishes. *Biological Oceanography* 5:215–29.

Trefry, J.H., S. Metz, T.A. Nelsen, R P. Trocine, and B.J. Eadie. 1994. Transport of particulate organic carbon by the Mississippi River and its fate in the Gulf of Mexico. *Estuaries* 17:839–49.

Turner, R.E. 1997. Wetland loss in the northern Gulf of Mexico: Multiple working hypotheses. *Estuaries* 20:1–13.

Turner, R.E. 1998a. A comparative mass balance budget (C, N, P and suspended solids) for a natural swamp and overland flow systems. In *Nutrient cycling and retention in natural and constructed wetlands*, ed. J. Vymazal, 61–71. Leiden, The Netherlands: Backhuys Publishers.

Turner, R.E. 1998b. Potential nutrient limitations on phytoplankton growth within and near the Louisiana Coastal Current. In *An observational study of the Mississippi–Atchafalaya coastal plume. Final report*, ed. S. P. Murray, 231–38. OCS Study MMS 98-0040. New Orleans, LA: U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region.

Turner, R.E., and R.L. Allen. 1982a. Bottom water oxygen concentration in the Mississippi River Delta Bight. *Contributions in Marine Science* 25:161–72.

Turner, R.E., and R.L. Allen. 1982b. Plankton respiration in the bottom waters of the Mississippi River Delat Bight. *Contributions in Marine Science* 25:173–79.

Turner, R.E., and N.N. Rabalais. 1991. Changes in Mississippi River water quality this century: Implications for coastal food webs. *BioScience* 41:140–48.

Turner, R.E., and N.N. Rabalais. 1994a. Changes in the Mississippi River nutrient supply and offshore silicate-based phytoplankton community responses. In *Changes in fluxes in estuaries: Implications from science to management*, ed. K.R. Dyer and R.J. Orth, 147–50. Proceedings of ECSA22/ERF Symposium, International Symposium Series. Fredensborg, Denmark: Olsen & Olsen.

Turner, R.E., and N.N. Rabalais. 1994b. Coastal eutrophication near the Mississippi River Delta. *Nature* 368:619–21.

Turner, R.E., and N.N. Rabalais. 1998a. Bottom water respiration rates in the hypoxia zone within the Louisiana Coastal Current. In *An observational study of the Mississippi–Atchafalaya coastal plume. Final report*, ed. S.P. Murray, 354–64. OCS Study MMS 98-0040. New Orleans, LA: U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region.

Turner, R.E., and N.N. Rabalais. 1998b. Light conditions within and near the Louisiana Coastal Current. In *An observational study of the Mississippi–Atchafalaya coastal plume. Final report*, ed. S.P. Murray, 199–207. OCS Study MMS 98-0040. New Orleans, LA: U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region.

Turner, R.E., W.W. Schroeder, and W.J. Wiseman, Jr. 1987. The role of stratification in the deoxygenation of Mobile Bay and adjacent shelf bottom waters. *Estuaries* 10:13–19.

Turner, R.E., N.N. Rabalais, and Z.-N. Zhang. 1990. Phytoplankton biomass, production and growth limitations on the Huanghe (Yellow River) continental shelf. *Continental Shelf Research* 10:545–71.

Turner, R.E., N.N. Rabalais, Q. Dortch, D. Justić, and B.K. Sen Gupta. 1997. Evidence for nutrient limitation and sources causing hypoxia on the Louisiana shelf. In *Proceedings of the first Gulf of Mexico hypoxia management conference, December 1995, New Orleans, Louisiana*, 106–12, Publication no. EPA-55-R-97-001, U.S. Environmental Protection Agency, Gulf of Mexico Program Office, John C. Stennis Space Center, MS.

Turner, R.E., N. Qureshi, N.N. Rabalais, Q. Dortch, D. Justić, R.F. Shaw, and J. Cope. 1998. Fluctuating silicate: Nitrate ratios and coastal plankton food webs. In *Proceedings of the National Academy of Science USA* 95:13048–51.

Turner, R.E., E.M. Swenson, M. Kasprzak, and T. Romaire. In review. LOOP/LDWF water quality analysis: Draft final report. Prepared for the Louisiana Transportation Research Center by the Coastal Ecology Institute, Louisiana State University, Baton Rouge, LA.

United Nations. 1951–1985. *Annual statistical yearbook*. New York, NY: United Nations

U.S. Army Corps of Engineers. 1974. Deep draft access to the ports of New Orleans and Baton Rouge. Draft environmental statement. New Orleans, LA: U.S. Army Corps of Engineers, New Orleans District.

U.S. Department of Agriculture (USDA). 1950–1985. *Annual reports: Agricultural statistics*. Washington, DC: Government Printing Office.

Valiela, I. 1984. *Marine ecological processes*. New York, NY: Springer-Verlag.

Walker, N.D. 1996. Satellite assessment of Mississippi River plume variability: Causes and predictability. *Remote Sensing of Environment* 58:21–35.

Walker, N.D., G.S. Gargion, L.J. Rouse, and D.C. Biggs. 1994. The great flood of summer 1993: Mississippi River discharge studied. *Eos, Transactions, American Geophysical Union* 75:409, 414–15.

Waller, R.S. 1998. Extent of hypoxia in areas east of the Mississippi River to Mobile Bay, 1998. Invited talk, Hypoxia Effects Workshop, March 11–13, Baton Rouge, LA.

Walsh, J.J. 1988. *On the nature of continental shelves*. New York, NY: Academic Press, Inc.

Walsh, J.J. 1991. Importance of continental margins in the marine biogeochemical cycling of carbon and nitrogen. *Nature* 350:53–55.

Ward, C.H., M.E. Bender, and D.J. Reish, eds. 1979. The offshore ecology investigation. Effects of oil drilling and production in a coastal environment. *Rice University Studies* 65:1–589.

Welsh, B.L., and F.C. Eller. 1991. Mechanisms controlling summertime oxygen depletion in western Long Island Sound. *Estuaries* 14:265–78.

Welsh, B.L., R.I. Welsh, and M.L. DiGiacomo–Cohen. 1994. Quantifying hypoxia and anoxia in Long Island Sound. In *Changes in fluxes in estuaries: Implications from science to management*, ed. K.R. Dyer and R.J. Orth, 131–37. Proceedings of ECSA22/ERF Symposium, International Symposium Series. Fredensborg, Denmark: Olsen & Olsen.

Whitledge, T.E. 1994. Nutrient concentrations at the Mississippi/Atchafalaya River outflows. In *Coastal oceanographic effects of summer 1993 Mississippi River flooding*, ed. M.J. Dowgiallo, 36–38. Special National Oceanic and Atmospheric Administration Report. Silver Spring, MD: NOAA, Coastal Ocean Office/National Weather Service.

Winger, P.V. 1986. *Forested wetlands of the Southeast: Review of major characteristics and roles in maintaining water quality*. U.S. Fish and Wildlife Service Resource Publication no. 13. Washington, DC: U.S. Department of the Interior, U.S. Fish and Wildlife Service.

Wiseman, W.J., Jr., and F.J. Kelly. 1994. Salinity variability within the Louisiana Coastal Current during the 1982 flood season. *Estuaries* 17:732–39.

Wiseman, W.J., Jr., J.M. Bane, S.P. Murray, and M.W. Tubman. 1975. Small-scale temperature and salinity structure west of the Mississippi River Delta. In *Memoires de la Societé Royale des Sciences de Liege, Belgium*, ed. J.C.J. Nihoul, X:277–85.

Wiseman, W. J., Jr., E.M. Swenson, and J. Power. 1991. Salinity trends in Louisiana estuaries. *Estuaries* 13:265–71.

Wiseman, W. J., Jr., V.J. Bierman, Jr., N.N. Rabalais, and R.E. Turner. 1992. Physical structure of the Louisiana shelf hypoxic region. In *Nutrient enhanced coastal ocean productivity*, 21–26. Publication no. TAMU-SG-92-109. Galveston, TX: Texas Sea Grant College Program.

Wiseman, W. J., Jr., N.N. Rabalais, R.E. Turner, S.P. Dinnel, and A. MacNaughton. 1997. Seasonal and interannual variability within the Louisiana Coastal Current: Stratification and hypoxia. *Journal of Marine Systems* 12:237–48.

Wright, L.D. 1970. Circulation, effluent diffusion and sediment transport, mouth of South Pass, Mississippi River Delta. Ph.D. diss., Department of Marine Sciences, Louisiana State University, Baton Rouge, LA.

Wulff, F., and L. Rahm. 1988. Long-term, seasonal and spatial variations of nitrogen, phosphorus and silicate in the Baltic: An overview. *Marine Environmental Research* 26:19–37.

Zucker, L.A., and L.C. Brown. 1998. *Agriculture drainage. Water quality impacts and subsurface drainage studies in the Midwest*. Ohio State University Extension Bulletin 871. Columbus, OH: The Ohio State University.

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