ESTUARINE SCIENCE

A Synthetic Approach to Research and Practice

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CHAPTER 2

Some Approaches for Assessing Human Influences on Fluxes of Nitrogen and Organic Carbon to Estuaries

Robert W. Howarth, Norbert Jaworski, Dennis Swaney,
Alan Townsend, and Gilles Billen

Abstract

Inputs of organic carbon and nitrogen to estuaries and coastal seas from nonpoint sources in the landscape are critically important in regulating the metabolism of these coastal marine ecosystems. This chapter summarizes two approaches we have used to investigate the controls on such fluxes. The first is the use of a model, the Generalized Watershed Loading Function (GWLF) model, to evaluate sources of organic carbon to the tidal, freshwater Hudson River estuary. This ecosystem is highly heterotrophic, and heterotrophic respiration is driven largely by allochthonous inputs of organic matter from nonpoint sources. GWLF suggests that the major source of this allochthonous carbon is erosion from agricultural fields. The Hudson Valley was once largely forested, reached a peak of agricultural activity in the early 1900s, and has steadily become more forested again as agricultural land has been abandoned. GWLF indicates that these land-use changes are likely to have resulted in large changes in the metabolism of the freshwater river, with the greatest heterotrophy coinciding with the greatest agricultural activity. The model also suggests that climate change may affect allochthonous inputs to the estuary, since drier soils result in substantially less erosion of agricultural soils.

The second approach we describe is a large-scale comparative analysis of nitrogen fluxes in rivers to the estuaries and coastal seas of the North Atlantic Ocean. For this analysis, we divided the watersheds of the North Atlantic Basin into fourteen large regions. Nonpoint sources of nitrogen dominate the flux for all regions. Nitrogen inputs to a region from human activity (import of food, use of inorganic nitrogen fertilizer, nitrogen fixation by agricultural crops, and atmospheric deposition of oxidized nitrogen originating from fossil-fuel combustion) are linearly related to nitrogen export to the coast from the region. On average, only 20% of the human-controlled inputs to a region are exported from the region to coastal waters, and the rest

are processed or retained in the landscape. Human activity has probably had substantial effects on increased nitrogen inputs to the coast for some regions, perhaps increasing inputs by some fifteen-fold for the North Sea and tenfold for the northeastern United States. Regression analysis suggests that the deposition of oxidized nitrogen from fossil fuel combustion may be particularly important as a source of nitrogen flowing from the landscape to estuaries; per unit nitrogen input, such deposition appears to contribute disproportionately to riverine flows in comparison to agricultural sources. Regression analysis also suggests that the deposition of ammonia and ammonium from the atmosphere may be a good surrogate measure of the leakage of nitrogen from agricultural systems to surface waters.

Introduction

The metabolism of estuaries is strongly influenced by the external inputs of organic carbon and nutrients (NRC 1993; Nixon 1995; Nixon et al. 1996; Kemp et al. 1997). In most estuaries, the majority of nutrient and organic matter inputs come from land, with upstream advection of nutrients from oceanic sources being relatively minor (Nixon et al. 1995, 1996; Howarth et al. 1995), although for some estuaries with limited watersheds the oceanic source can dominate metabolism (Smith et al. 1991). Despite the obvious importance of fluxes of nitrogen, phosphorus, and organic carbon from the landscape to estuaries, there has been relatively little study on the control of these fluxes and surprisingly few syntheses (but see Jaworski 1997; Meybeck 1982; Billen et al. 1991; Hopkinson and Vallino 1995). While sewage inputs dominate in some estuaries, nonpoint-source fluxes are often greater (Nixon and Pilson 1983; NRC 1993; Howarth et al. 1996a, 1996b). Much evidence points to increased inputs of nutrients and organic carbon to estuaries from human activity in the landscape (Meybeck 1982; Pacés 1982; Larsson et al. 1985; Turner and Rabalais 1991; Howarth et al. 1996a, 1996b). In this chapter, we summarize two approaches we have used to estimate the extent of this human influence and to predict future changes: (1) a modeling approach for examining sources of organic matter input to the Hudson River estuary; and (2) a comparative regional analysis of nitrogen fluxes from land to the estuaries and coastal seas of the North Atlantic Ocean.

Modeling Sources of Organic Matter Input to the Hudson River

For the past decade, we have been studying the metabolism of the Hudson River estuary. The tidal, freshwater Hudson River (a stretch of river approximately 150 km long) is highly heterotrophic, with ecosystem respiration exceeding gross primary production (GPP) by almost twofold (Howarth et al. 1992, 1996a). This heterotrophic metabolism is fueled largely by organic carbon inputs to the river from nonpoint sources in the landscape; these nonpoint inputs exceed inputs of organic carbon in sewage by tenfold (Howarth et al. 1996a). Many other large rivers and estuaries also receive significant inputs of allochthonous organic matter from their watersheds (Schlesinger and Melack 1981; Meybeck 1982; Richey et al. 1991), and many other river and estuarine ecosystems are also heterotrophic (Kempe et al. 1991; Smith et al. 1991; Smith and Hollibaugh 1993; Hopkinson and Vallino 1995). These observations lead to some interesting questions, such as: To what extent are these fluxes of organic carbon from the landscape natural, and to what extent are they influenced by human activity? Is there an important climatic control on carbon inputs to rivers and estuaries, and how may climate change affect organic carbon fluxes from the landscape? To address these questions, we have adapted the GWLF model to estimate fluxes of organic carbon to the Hudson River from its watershed (Howarth et al. 1991; Swaney et al. 1996). GWLF was originally developed by Haith and Shoemaker (1987) to estimate fluxes of nitrogen and phosphorus from watersheds of mixed land use in the northeastern United States.

The GWLF model as we have used it relies on a simple mass-balance approach to estimate surface hydrologic and groundwater fluxes of water (figure 2-1). Daily climatic data from NOAA weather stations drive the hydrologic part of the model; precipitation is divided into rainfall or snow based on temperature; melting of the snowpack, if any, is estimated from temperature; and evapotranspiration is estimated from temperature and the number of daylight hours. Groundwater is treated in two reservoirs: a surface zone which is usually not saturated and a deeper, permanently saturated zone; water is added to the surface zone from rainfall and snowmelt and is removed through evapotranspiration and percolation into the lower saturated zone. Percolation is a first-order function of the water content in the surface zone; when the surface zone becomes saturated, any further rainfall and snowmelt becomes surface runoff. Groundwater movement from the lower saturated zone to streams and rivers is a first-order function of the water content in the lower zone. In our first effort with the model, we used spatially averaged daily weather inputs for the entire Hudson River watershed for the meteorological inputs (Howarth et al. 1991); Swaney et al. (1996) improved on this approach by dividing the Hudson watershed into eight sub-basins and using averaged daily weather inputs for each sub-basin.

The GWLF model uses the universal soil loss equation (Stewart et al. 1976) to estimate erosion from forests and agricultural lands and the STORM model (Hydraulic Engineering Center 1977) to estimate erosion from urban and suburban areas (figures 2-2 and 2-3). For urban and suburban areas, erosion is a relatively simple function of the surface runoff of water estimated from

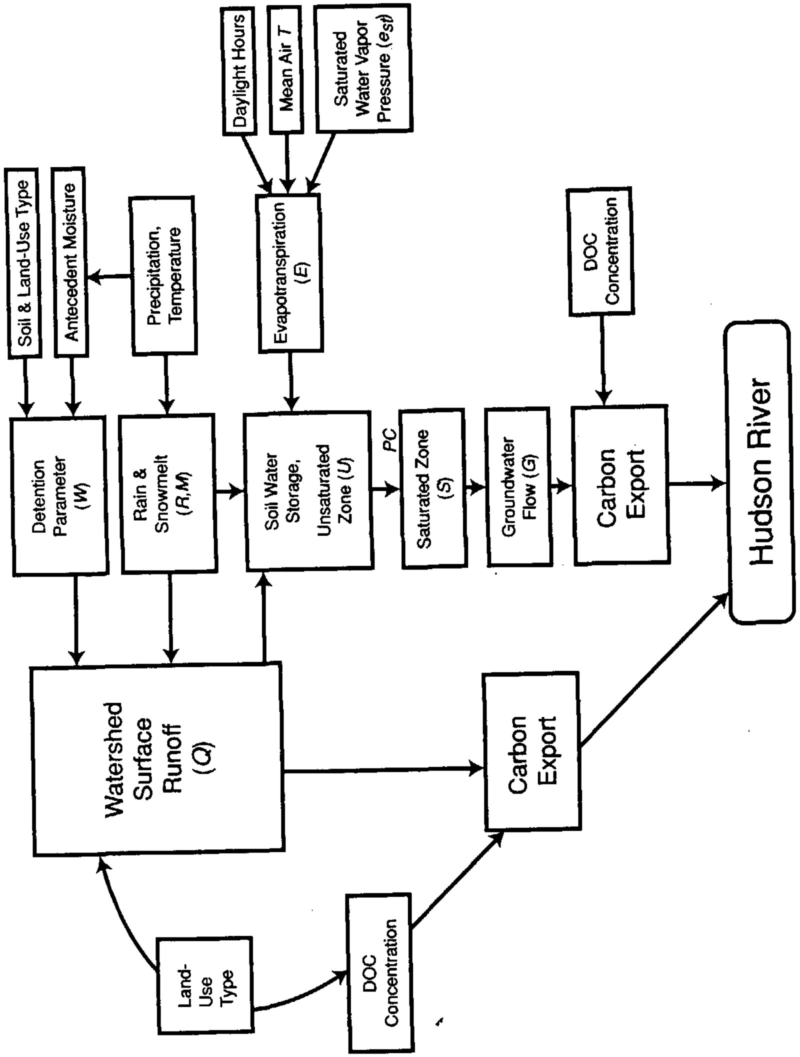

FIGURE 2-1 Structure for the hydrological portion of the GWLF model. Concentrations of dissolved organic carbon (DOC) are assigned to groundwater flows and to surface-water runoff as a function of land use. Reprinted from Howarth et al. (1991) by permission.

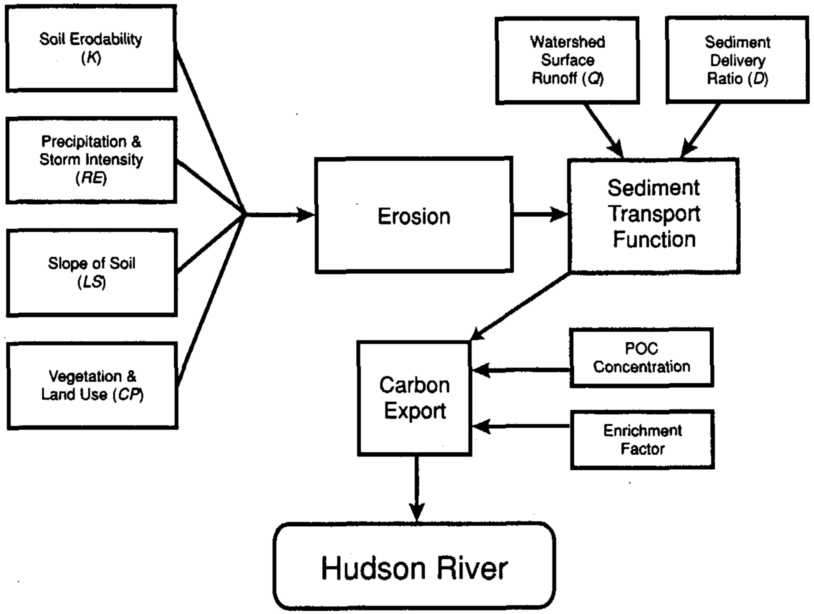

FIGURE 2-2 Structure of the GWLF model for the movement of sediment in forests and agricultural systems. Structure is based on the universal soil-loss equation. The concentration of particulate organic carbon is assigned to eroded soils as a function of land use. Reprinted from Howarth et al. (1991) by permission.

GWLF's hydrologic submodel; all soil eroded from urban and suburban lands is assumed to reach surface waters. For forests and agricultural lands, erosion is treated as a function of precipitation and intensity of storms, the type of land use, the erodability of soil, and the slope of the land. The model assumes that only a small amount of the soil eroded from forest and fields actually reaches surface waters, while most is retained in the landscape. The proportion that reaches surface waters is estimated from the "sediment delivery ratio," which in turn is estimated from the area of the watershed; the larger the watershed, the smaller the sediment delivery ratio. The transport of eroded soils to surface waters is estimated on a monthly basis using a power function of the surface runoff estimate derived from GWLF's hydrologic submodel (Haith and Shoemaker 1987; Howarth et al. 1991; Swaney et al. 1996).

Although GWLF was developed as a tool to estimate fluxes of nitrogen and phosphorus to surface waters (Haith and Shoemaker 1987), the model has no explicit inclusion of biogeochemical processes. Rather, concentrations of dissolved and particulate nutrients (Haith and Shoemaker 1987) or concentrations of dissolved and particulate organic carbon (Howarth et al. 1991) are

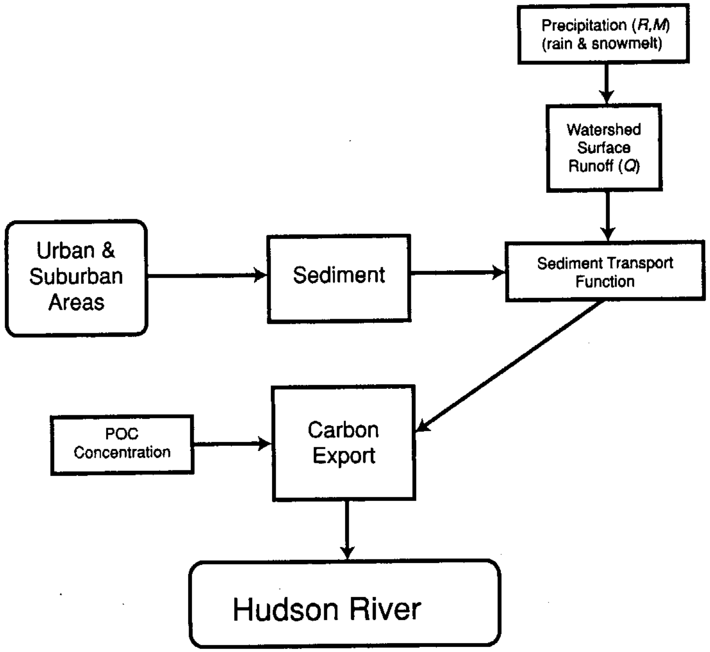

FIGURE 2-3 Structure of the GWLF model for movement of sediment in urban and suburban areas. Reprinted from Howarth et al. (1991) by permission.

assigned to flows of surface water, groundwater, and sediments as a function of land-use type. Also, although actual land-use data are used in the GWLF model, there is no spatial reality to the model and land areas are aggregated. That is, the model has no explicit recognition to whether water leaving agricultural fields flows through riparian wetlands or goes directly into surface waters.

The GWLF model does a very good job of estimating freshwater discharge and a good job of estimating fluxes of sediment to the Hudson River, particularly when using meteorological inputs at the finer scale of eight sub-basins (Swaney et al. 1996). Compared with freshwater discharge and sediment flux estimates based on U.S. Geological Survey (USGS) data from the Green Island gauging station, which includes the upper Hudson and Mohawk Rivers (two tributaries which together comprise two-thirds of the freshwater discharge for the entire Hudson River), the model underestimates average annual discharge by 9% and average annual sediment flux by 34% (Swaney et al. 1996). The model also does a good job of catching the seasonal variability in both freshwater discharge and sediment flux over a 3-year period (figures 2-4 and 2-5).

Evaluating the GWLF-derived estimate for organic carbon input to the Hudson River estuary is more difficult, as organic carbon fluxes have not been

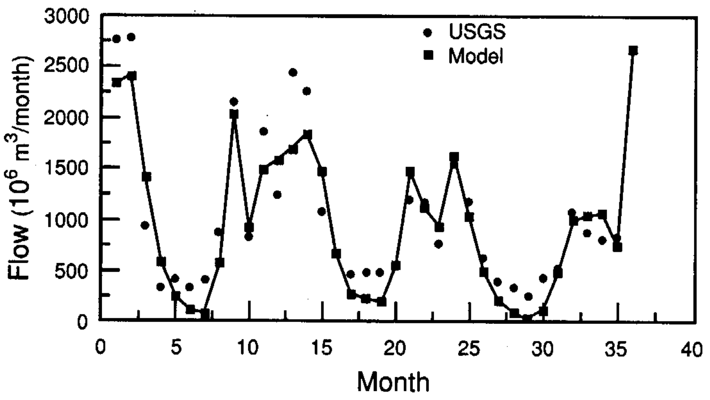

FIGURE 2-4 GWLF model output for freshwater discharge for the upper Hudson and Mohawk Rivers compared with data from the USGS gauging station at Green Island. The model run is for 36 months beginning in April 1983. USGS data are solid points. Model output is indicated by the line. Reprinted from Swaney et al. (1996) by permission.

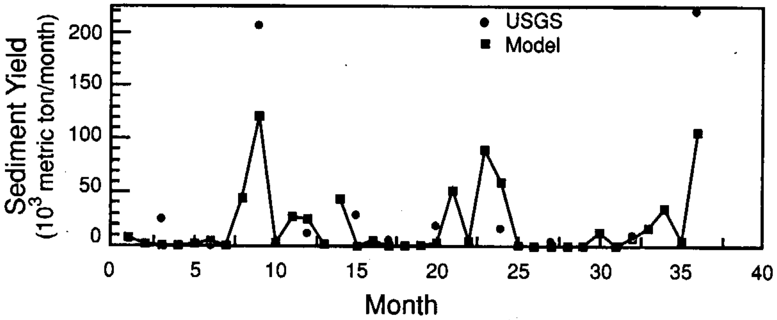

FIGURE 2-5 GWLF model output for sediment fluxes for the upper Hudson and Mohawk Rivers compared with data from the USGS gauging station at Green Island. The model run is for 36 months beginning in April 1983. USGS data are solid points. Model output is indicated by the line. Reprinted from Swaney et al. (1996) by permission.

routinely measured at the Green Island gauging station by the USGS. However, the USGS has occasionally measured organic carbon in many tributaries within the Hudson River watershed. Gladden et al. (1988) regressed the log of the flux of organic carbon per watershed area against the log of water discharge per watershed area for all times and places within the Hudson River watershed for which data were available for the early 1980s. Using this regression and estimates for total freshwater discharge, Gladden et al. (1988) estimated the flux of organic carbon from the watershed of the Hudson to be

3.1 g C m^{-2} yr^{-1} for an average year with typical freshwater discharge. A mass balance of organic carbon in the freshwater Hudson River for 1988 gives great credibility to this approach (Howarth et al. 1996a); 1988 was a drought year with low freshwater discharge; for that year, the approach of Gladden et al. (1988) yields an estimate of 2.4 g C m^{-2} yr^{-1} for allochthonous inputs of organic carbon to the Hudson; summing this estimate with other inputs gives a total estimate of carbon inputs to the Hudson that agrees to within 2% with the sum of carbon sinks for that same year (Howarth et al. 1996a). The years we modeled with GWLF had average precipitation, and so our best estimate of allochthonous inputs of organic carbon to the Hudson from the landscape for these years would be 3.1 g C m^{-2} yr^{-1} (Howarth et al. 1996a). Compared with this, the GWLF model estimate is low by 44% (Swaney et al. 1996; see also table 2-1). That GWLF underestimates organic carbon fluxes to a greater extent than total sediment fluxes may be due to the lack of consideration of in-stream and in-river processing of organic carbon by the model (Howarth et al. 1991).

The GWLF model estimates that in the Hudson watershed agricultural lands export tenfold more organic carbon per area than do forests with urban and suburban lands exporting an intermediate amount of carbon (table 2-1). The Hudson River watershed is largely forested (65% of total area), and only 28% of the land is in agricultural use, yet the GWLF model suggests that agricultural lands contribute 74% and forested lands only 18% of the total inputs of organic carbon to the Hudson from nonpoint sources (Swaney et al. 1996). Urban and suburban lands, which comprise 7% of the area, are estimated to contribute 8% of the flux of nonpoint-source organic carbon (Swaney et al. 1996).

TABLE 2-1

Estimates of inputs of organic matter from nonpoint sources in the watershed to the tidal, freshwater Hudson River.

	g C m^{-2} yr^{-1}
Regression analysis—average for all of Hudson River	3.1
Model result—average for all land uses	1.7
Model result—forest lands only	0.47
Model result—agricultural lands only	4.5
Model result—urban and suburban lands only	1.9

The regression analysis is based on Gladden et al. (1988) and uses average freshwater discharge for the Hudson River and a regression of the log of water discharge per area vs. the log of organic carbon export per area for various tributaries of the Hudson (Howarth et al. 1996a). The model-derived estimates are from runs of GWLF reported by Swaney et al. (1996). All estimates are for years of average precipitation and discharge.

If these estimates of organic carbon input are accurate, then carbon loading to the Hudson River has changed dramatically as land use has changed. Before European settlement, the Hudson River watershed was largely forested; running GWLF with 1980s climatic data but assuming that 100% of the land is forested, the model estimates that organic carbon inputs would be only one-third of modern values (Swaney et al. 1996). This suggests that in its pristine state, the Hudson River would have been much less heterotrophic than at present (Howarth et al. 1996a). On the other hand, in the early 1900s most of the land in the Hudson River watershed was actively farmed, and the current situation of 65% forest use is a result of gradual agricultural abandonment and forest regrowth (Rod et al. 1989). Running GWLF with 1980s climatic data but assuming land use as it existed at peak agricultural use, the model estimates that nonpoint loading of organic carbon to the Hudson may once have been almost twice as large as at present (Swaney et al. 1996); the actual situation may have been even worse since we assumed modern agricultural practices for the model run but actual practices probably resulted in more erosion. Thus, the Hudson was probably more heterotrophic earlier in the 1900s than at present (Howarth et al. 1996a). This conclusion should perhaps be tempered because of changes in sewage inputs over time due to population growth and changes in sewage treatment technology; while sewage inputs are currently only 10% of the nonpoint source allochthonous inputs of carbon from the landscape (Howarth et al. 1996a) and are unlikely ever to have been the major input, the organic carbon in sewage is more labile and more easily respired than is the carbon in allochthonous inputs from eroded soils. Nonetheless, heterotrophic respiration in the freshwater Hudson currently is driven largely by the allochthonous inputs from nonpoint sources.

Gladden et al. (1988) noted that year-to-year variability in climate probably results in substantial interannual variability in organic carbon loading to the Hudson River. Hence, climate change could have a profound influence on the metabolism of the Hudson and other rivers and estuaries in the northeastern United States; models of global warming suggest a generally drier climate for the northeastern United States, which might be expected to decrease inputs of organic carbon from land to these ecosystems (Moore et al. 1997). Our runs with the GWLF model, however, indicate that the most important climatic factors regulating organic carbon fluxes from the landscape are not average annual discharge or precipitation, but rather day-to-day and seasonal patterns of precipitation and discharge (Howarth et al. 1991). Thus, even if climate change results in a generally drier climate, allochthonous fluxes of organic carbon might increase in years with periods of intense summer storms.

The GWLF model has contributed greatly to our understanding of carbon flows through the Hudson River watershed. GWLF has also been used to explore sources of nitrogen to tributaries of the Delaware River (Haith and

Shoemaker 1987). However, as noted above, the model relies entirely on hydrology, erosion, and concentrations of carbon and nutrients assigned as functions of land use, and does not contain any explicit representation of biogeochemical processes. We believe strongly that the inclusion of biogeochemical processes—both in-stream and in the terrestrial ecosystems—is necessary to make models more useful in exploring sources of nutrients and organic carbon for rivers and estuaries. We are aware of no model that has yet linked biogeochemical processes in terrestrial systems with hydrologic and erosional processes, and with in-stream processing of organic carbon or nutrients, but such linkages should be possible. The Century model has proven quite powerful and versatile for studying a variety of carbon and nitrogen biogeochemical processes in terrestrial ecosystems (Parton et al. 1987, 1996; Schimel et al. 1994) and shows promise as a tool for understanding export to aquatic ecosystems (Baron et al. 1994). Other terrestrial biogeochemical models have been developed which may be even more useful for modeling nitrogen fluxes from agricultural systems (Scholefield et al. 1991; Engel and Priesack 1993; Grant et al. 1993; Müller et al. 1997). We believe that linking a terrestrial biogeochemical model and a hydrological/erosion model such as GWLF could prove quite powerful in predicting fluxes of both carbon and nitrogen from the landscape. Further refinements could include processing of materials in riparian wetlands and in streams and rivers. For example, in-stream processing of nitrogen can perhaps be modeled simply by estimating retention as a function of water depth and residence time (Kelly et al. 1987; Howarth et al. 1996b). Similarly, in-stream processing of organic carbon can perhaps be modeled from information on "spiraling distances" or turnover times in streams and rivers of different sizes (Newbold 1992; Howarth et al. 1996a).

A Comparative Regional Analysis of Nitrogen Fluxes from Land to the North Atlantic Ocean

Nitrogen is frequently the element most limiting to primary productivity in estuaries and coastal seas in the temperate zone, and the management of coastal eutrophication demands that nitrogen inputs to estuaries be better controlled (Howarth 1988; NRC 1993; Nixon 1995; Nixon et al. 1996; Downing 1997). In many estuaries, nonpoint sources dominate inputs of nitrogen (Nixon and Pilson 1983; NRC 1993), and for the North Atlantic Ocean as a whole, nonpoint-source fluxes of nitrogen from the landscape exceed sewage inputs by ninefold (Howarth et al. 1996b). Thus, better management of coastal eutrophication requires an understanding of the nonpoint sources of nitrogen to surface waters from the surrounding landscape.

Anthropogenic activity has greatly accelerated nitrogen cycling and has caused the rate of nitrogen fixation in the terrestrial biosphere to at least double over natural biological rates (Galloway et al. 1995; Vitousek et al. 1997). This increase in nitrogen fixation, which is due largely to the manufacturing of inorganic nitrogen fertilizer but also includes substantial incidental nitrogen fixation from the combustion of fossil fuels, has been dramatic over the past few decades and continues to increase at an exponential rate (Vitousek et al. 1997). One should perhaps expect that the increased nitrogen fixation on land might result in larger fluxes of nitrogen to estuaries, and indeed several papers have reported increased fluxes of inorganic nitrogen as nitrate in rivers over time. For the Mississippi River, nitrate concentrations have more than doubled in the last 30 years (Turner and Rabalais 1991). Nitrate concentrations in many rivers of Central Europe (Pacés 1982) and the northeastern United States (Jaworski et al. 1997) have increased four- to tenfold during the 1900s.

While it is reasonable to deduce that both the increased use of inorganic fertilizer and increased atmospheric deposition of nitrogen from fossil-fuel combustion contribute to elevated riverine fluxes of nitrogen to the coast, estimating the relative contribution of each of these sources is a difficult challenge (Howarth et al. 1996b). Consider the relatively simple case of determining the fate of inorganic nitrogen fertilizer. If the fertilizer is applied to a grassland on a loam soil, only 3 to 10% of the application typically is leached as nitrate, but if the fertilizer is applied to tilled cropland on a sandy soil, from 25 to 80% of the fertilizer is generally leached (Howarth et al. 1996b). Even in the absence of fertilization, converting pastureland to tilled cropland appears to greatly increase nitrogen leaching, and the extent of this leaching varies with soil type (Howarth et al. 1996b). Despite this complexity, the general patterns of nitrogen leaching from agricultural systems are relatively well understood, and estimates of such leaching can be made for large regions (Howarth et al. 1996b). However, leaching of nitrate from agricultural fields is not the only pathway whereby fertilizer can reach surface waters; significant amounts can be volatilized as ammonia, with the ammonium later being deposited back onto the landscape where it may be nitrified and leached. Also, one must consider the fate of nitrogen in harvested crops; on average, about half of nitrogen fertilizer applied to tilled croplands is removed in the crop when it is harvested (Bock 1984; Nelson 1985). Much of the crop is then fed to animals, which can result in significant volatilization of nitrogen to the atmosphere or direct leaching of nitrogen to surface waters. Thus, to determine the fate of nitrogen from added fertilizer, the correct unit of study is not agricultural field plots but rather the entire food-agricultural system (Bleken and Bakken 1997a, 1997b; Isermann and Isermann 1998). At this scale, the linkage between fertilizer use and nitrogen inputs to surface waters remains poorly documented.

Estimating the fate of nitrogen in atmospheric deposition onto the landscape is at least as challenging as determining the fate of nitrogen fertilizer. Consider deposition onto forests. Ecological theory would suggest that forest age is a major determinant of the fate of nitrogen deposition, with young aggrading forests retaining nitrogen and old-growth forests with zero rates of net ecosystem production exporting nitrogen (Vitousek and Reiners 1975; Hedin et al. 1995). However, even young, aggrading, nitrogen-limited forests can export large amounts of nitrogen from atmospheric deposition; factors such as highly permeable, sandy soils and nitrogen loss during a winter dormant season can contribute to nitrogen loss from young forests (Lajtha et al. 1995). Most forests of the temperate zone are nitrogen limited (Vitousek and Howarth 1991), but at high rates of atmospheric deposition, nitrogen inputs can become sufficient to "saturate" nutritional needs and other elements may become limiting. Under these conditions of nitrogen saturation, nitrogen leaching to downstream aquatic ecosystems can greatly increase (van Breemen et al. 1982; Aber et al. 1989; Schulze 1989; Gundersen and Bashkin 1994). This linkage between nitrogen deposition, nitrogen saturation of an ecosystem, and nitrogen export has received a tremendous amount of study, yet the relationship between nitrogen inputs and nitrogen export from forests remains unclear. In a review of data for forests in the United States, Johnson (1992) found no relationship between nitrogen inputs and nitrogen exports; however, few if any of the studies reviewed by Johnson included export of organic nitrogen; organic nitrogen export can be large and if included might result in clearer relationships between input and export (Hedin et al. 1995; Howarth et al. 1996b). For European forests, nitrogen leaching tends to be either quite small or quite large (Hauhs et al. 1989). Recently, Gundersen et al. (1997) have demonstrated that nitrogen export from European forests, while not correlated with nitrogen deposition, is well correlated with variables indicative of the nitrogen status in forests, such as the C:N ratio of the forest floor.

Much progress is being made in understanding the cycling of nitrogen at the scale of individual ecosystems, yet given the complexity of nitrogen cycling at this scale, the uncertainty in scaling up nitrogen export from individual ecosystems to larger scales is likely to remain great. To determine if more predictability in nitrogen export from the landscape can be gained from analysis at larger spatial scales, we undertook an analysis of nitrogen flux from land to the North Atlantic Ocean at the scale of large regions (Howarth et al. 1996b). We divided up the watersheds of the North Atlantic Ocean into fourteen large geographical areas; for each region, we estimated nitrogen flux in rivers and in sewage to estuaries and coastal seas. We also estimated the inputs of nitrogen from human activity into each region. This allowed us to perform a mass balance of human-derived nitrogen for each region. This effort was part of the first activity of the International SCOPE Nitrogen Project: an evaluation of the nitrogen cycle of the North Atlantic Ocean (Howarth 1996; Galloway et al. 1996). The International SCOPE Nitrogen Project was established to assess how humans have altered the nitrogen cycle at regional and global scales, and to determine the consequences of this alteration.

In our regional analysis, we used the best available data for each region and therefore used a variety of approaches for estimating the export of nitrogen in rivers to the coast. For many regions, good data on actual riverine nitrogen flows exist. This is particularly true for large regions dominated by a single river, as in the Amazon and Mississippi River Basins. For the northeastern United States, the southwestern coast of Europe, and the watersheds of the North Sea, nitrogen flows from the landscape to the coast occur in numerous rivers, but good data on nitrogen flows exist for most of these rivers (Howarth et al. 1996b). For some regions where data for individual rivers are scarce, we used hydrologic budgets and extrapolation of nitrogen concentrations from some smaller watersheds within the region. For some other regions, we used previously published estimates for regional nitrate export and estimated organic nitrogen export from estimates of organic carbon export and assumed C:N ratios (Howarth et al. 1996b).

The flux of nitrogen in rivers and sewage to the coastal areas of the North Atlantic Basin varies some twentyfold among the regions (Howarth et al. 1996b). The highest fluxes are in the regions with the highest population densities such as the North Sea area, northwestern Europe, and the northeastern United States. Peierls et al. (1991) reported that the log of nitrate fluxes from large rivers of the world are correlated with the log of population densities in their watersheds, and the same relationship holds for the North Atlantic regional analysis of total nitrogen fluxes, with the log of total nitrogen fluxes per area correlated with the log of population densities (Howarth et al. 1996b). When expressed on a linear-linear rather than a log-log scale, population density is even a better predictor of total nitrogen export from regions of the North Atlantic Basin ($r^2 = 0.77$, as opposed to 0.45 for the log-log relationship; Howarth et al. 1996b).

From the analysis of Peierls et al. (1991) for large rivers of the world, Cole et al. (1993) suggested that population density is correlated with nitrate fluxes because sewage is the major source of nitrogen to rivers in regions with higher population densities. However, sewage alone cannot explain the pattern between population density and total nitrogen fluxes for the regions of the North Atlantic Basin. For our analysis, we estimated the contribution of human sewage to nitrogen export from each region, using both population-based estimates and data on the extent of sewers in each region. Sewage is not the major contributor to riverine export of total nitrogen for any of the regions, and nonpoint sources of nitrogen dominated the riverine nitrogen flux in all regions surrounding the North Atlantic Ocean (Howarth et al.

1996b). Sewage is most important in the watersheds of the North Sea, where it contributes 34% of the riverine nitrogen flux, and is least important in the Amazon River Basin, where population density is very low and sewage contributes only 0.01% to the riverine nitrogen flux. On average, sewage contributes some 11% of the total nitrogen flux from land to the North Atlantic Ocean.

To analyze mass balances of human-derived nitrogen for each of the regions of the North Atlantic Basin, we considered the following as inputs of nitrogen to a region: the use of inorganic nitrogen fertilizer, nitrogen fixation by agricultural crops, the net movement of nitrogen into or out of a region in food and feedstocks, and the deposition of oxidized nitrogen compounds from the atmosphere (NO_y). To avoid double accounting, we did not consider sewage or the use of organic manure fertilizers as inputs of nitrogen to a region; both sewage and manure can contribute substantially to nitrogen inputs to surface waters, but the sewage and manure are a recycling of nitrogen within a region and not a new input. The nitrogen in the sewage and manure comes from regional inputs of nitrogen such as import of food/feedstocks or use of inorganic fertilizer to grow the food and feedstocks within the region which are then eaten by humans or livestock. Similarly, we did not consider the atmospheric deposition of ammonia/ammonium as an input to a region. Since most ammonia moves only fairly short distances through the atmosphere, ammonia deposition largely reflects nitrogen recycling within a region rather than a new input (Howarth et al. 1996b). The majority of the ammonia/ammonium from atmospheric deposition originated as volatilization of ammonia within the same region, and came from sources such as manure and feedlots; at the scale of large regions, the original source of the nitrogen in this ammonia is from the import of food, the use of inorganic fertilizer, nitrogen fixation by crops, or the deposition of oxidized nitrogen. In contrast to ammonia/ammonium deposition, the deposition of oxidized nitrogen (NO_y) to a region needs to be considered a net input of nitrogen to the region as most comes from a source not otherwise accounted for: the combustion of fossil fuels (Howarth et al. 1996b).

Sources of data for nitrogen inputs to a region are described in detail in Howarth et al. (1996b). Estimates for NO_y deposition due to human activity were derived from models and include a subtraction of background natural levels for each region from the estimate of current deposition (Prospero et al. 1996; Howarth et al. 1996b). For most regions, inorganic fertilizer is the largest single input of nitrogen, but NO_y deposition dominates in the northeastern United States and in the St. Lawrence Basin (Howarth et al. 1996b). For the regions as a whole, inorganic nitrogen fertilizer is the largest input and makes up almost two-thirds of all inputs (table 2-2).

TABLE 2-2

Anthropogenic nitrogen budgets per area of land for the northeastern United States and for all temperate watersheds of the North Atlantic Basin (kg N km^{-2} yr^{-1}). Data are from Howarth et al. (1996b).

	Northeastern United States	All Temperate Watersheds of North Atlantic Ocean
Inputs		
NO_y deposition	1,200	490
N fertilizer	600	1,400
N fixation in agriculture	750	315
Net import of N in food	1,000	0
Total inputs	3,550	2,205
Sinks and exports		
Net export of N in food	0	255
Export in rivers and sewage	1,070	490
Storage in the landscape and denitrification (by difference)	2,480	1,460
Total sinks and exports	3,550	2,205
Agricultural and wastewater fluxes within regions		
Wastewater N flux	280	80
Volatilization and deposition of NH_x	190	240

Per area, the total input of nitrogen from human activity varies more than thirtyfold among regions (Howarth et al. 1996b). Total net inputs (where export of nitrogen in food/feedstocks from a region is taken as a negative input) of human-controlled nitrogen are almost twice as great for the North Sea region as for any other region when expressed per land area. The lowest inputs per area are to the northern Canada region and to the Amazon River Basin (Howarth et al. 1996b). For the temperate regions of the North Atlantic Basin, the riverine nitrogen export per area of landscape is well correlated with the sum of the human-controlled inputs of nitrogen (figure 2-6). Compared to the temperate regions, tropical areas such as the Amazon River Basin export somewhat more nitrogen per unit of human-controlled nitrogen input to a region (Howarth et al. 1996b), perhaps because the terrestrial biosphere of the tropics is less nitrogen limited than in the temperate zone (Vitousek and Howarth 1991). The tropical regions of the North Atlantic Basin are not included in figure 2-6.

For the regression of riverine nitrogen export on human-controlled nitrogen inputs to regions, the *y*-intercept suggests a riverine nitrogen export of approximately 100 kg N km^{-2} yr^{-1} in the absence of human inputs of nitrogen to a region. The 95% confidence interval around the intercept is large (from a negative value to +400 kg N km^{-2} yr^{-1}), but an export of approximately 100 kg N

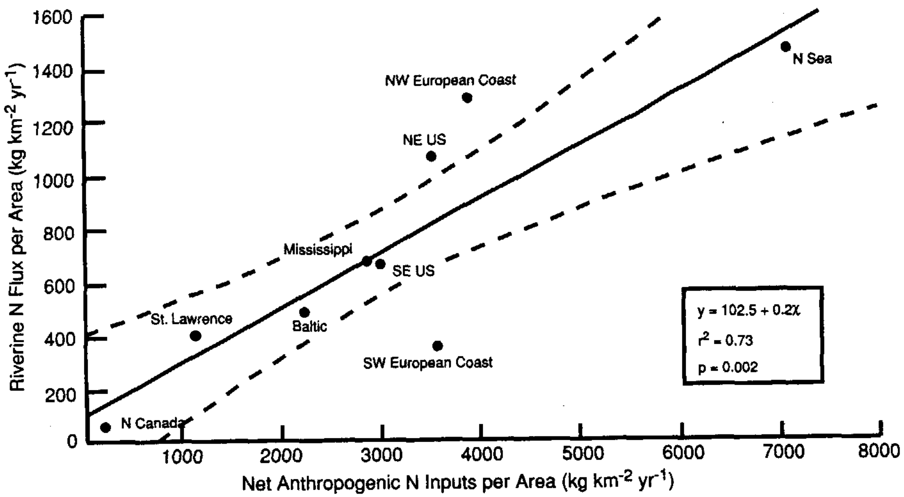

FIGURE 2-6 Riverine nitrogen export per area as a function of human-controlled nitrogen inputs for the temperate-zone regions of the North Atlantic Basin. Reprinted from Howarth et al. (1996b) by permission.

km^{-2} yr^{-1} from pristine temperate-zone systems is not far from other estimates for fluxes from pristine temperate systems compiled by Howarth et al. (1996b): 75 to 230 kg N km^{-2} yr^{-1}. The intercept-based estimate for pristine nitrogen exports suggests that current fluxes from the northeastern United States and from the North Sea region may be tenfold and from fifteenfold greater, respectively, than would be the case absent human activity. The slope of the regression illustrates that on average only 20% of the human-controlled inputs of nitrogen are exported to estuaries and the coast. The rest of this nitrogen must be processed or retained in the landscape; we suspect that most of this "missing" nitrogen is denitrified (Howarth et al. 1996b).

Atmospheric deposition of NO_y may be proportionately more important as a source of nitrogen to rivers and the coast than are the other nitrogen inputs to regions; multiple regression analysis suggests that per unit nitrogen input per area, NO_y deposition is some sevenfold better at predicting riverine nitrogen flows than are agricultural sources of nitrogen (Howarth et al. 1996b; Howarth 1998). Even though NO_y deposition is the largest nitrogen input to only two regions, it alone is a very good predictor of riverine nitrogen export from the temperate-zone regions of the North Atlantic Basin (figure 2-7). Inorganic nitrogen fertilizer is the largest nitrogen input to all of the other temperate-zone regions of the North Atlantic Basin and makes up two-thirds of the total inputs on average for all the regions (table 2-2); while nitrogen fertilizer is significantly correlated with riverine nitrogen export, it is far less predictive than NO_y deposition for riverine flux (figure 2-8). This suggests the importance of controlling fossil-fuel sources of atmospheric nitrogen pollution, particularly in the northeastern United States, where these are the dominant nitrogen inputs to the region.

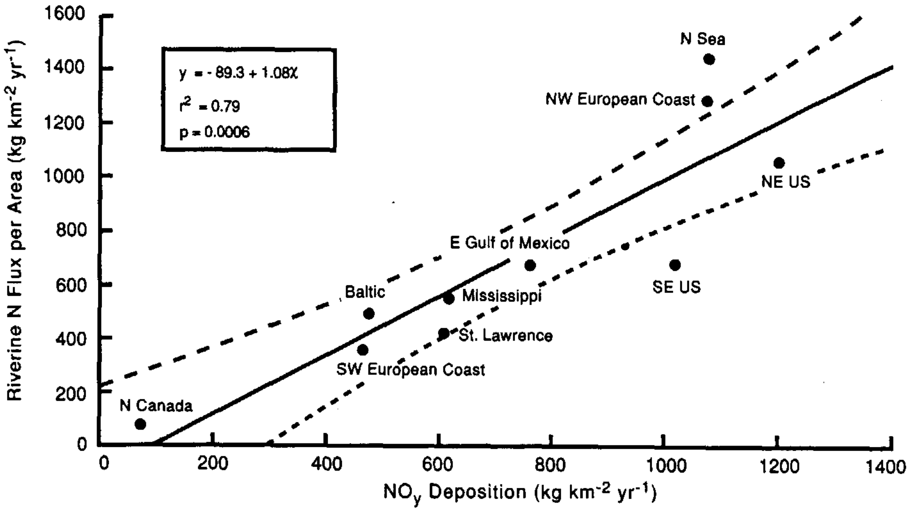

FIGURE 2-7 Riverine nitrogen export per area as a function of NO_y deposition for the temperate-zone regions of the North Atlantic Basin. Most of the nitrogen in NO_y deposition originates in the combustion of fossil fuels. Reprinted from Howarth et al. (1996b) by permission.

Even though inorganic fertilizer use is only a fair predictor of the export of nitrogen from a region, agricultural sources and practices clearly affect nitrogen export. Some recent analyses suggest that the major environmental impacts from nitrogen in agricultural systems come from "surplus" nitrogen that results in "leaky spots" (Bleken and Bakken 1997a, 1997b; Isermann and Isermann 1998). At the scale of the large regions of the North Atlantic Basin, the deposition of ammonia/ammonium may provide a good estimate of nitrogen flows into surface waters from these leaky spots in the agricultural system, as indicated by regressing ammonia/ammonium deposition plus NO_y deposition against riverine nitrogen fluxes from nonpoint sources (Howarth 1998). The nonpoint-source component of the riverine nitrogen flux for the regions of the North Atlantic Basin can be estimated by subtracting the estimated sewage input from the total riverine flux estimate for each region (Howarth 1998). We include NO_y deposition in the analysis because of its demonstrated relationship to riverine nitrogen export from regions. Estimates for ammonia/ammonium deposition are from Howarth et al. (1996b) and are

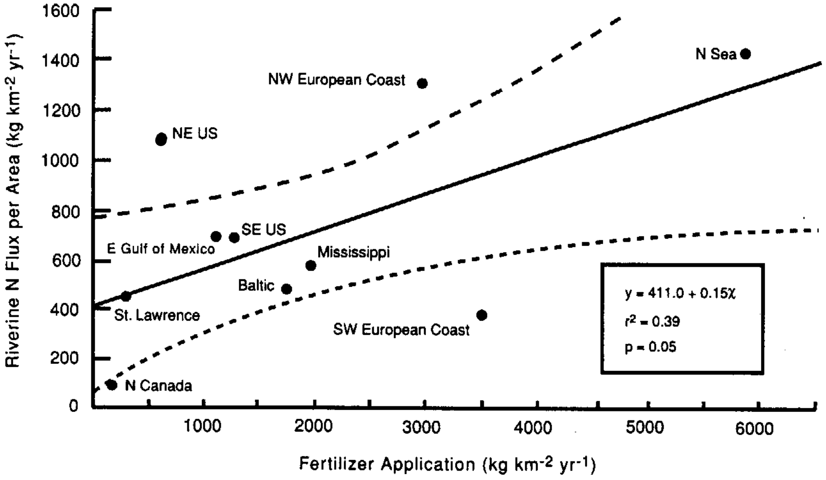

FIGURE 2-8 Riverine nitrogen export per area as a function of use of inorganic nitrogen fertilizer for the temperate-zone regions of the North Atlantic Basin. Reprinted from Howarth et al. (1996b) by permission.

based on models of emission strength and atmospheric transport. The regression of total inorganic nitrogen deposition, including wet and dry deposition of ammonia and ammonium, against the nonpoint-source flux of nitrogen from regions is an astonishingly good fit ($r^2 = 0.93$; figure 2-9).

Note that this analysis of total inorganic nitrogen deposition differs from the mass-balance approach we described above; ammonia/ammonium deposition is not an input of nitrogen to the region, but rather a recycling of nitrogen primarily from agricultural sources within the region (Schlesinger and Hartley 1991; Howarth et al. 1996b). However, ammonia/ammonium deposition appears to be a good measure of the agricultural component of nitrogen in a region that contributes to riverine fluxes to the coast. We stress that the regression shown in figure 2-9 need not indicate that ammonia/ammonium deposition is itself a major source of nitrogen to the riverine flux. An alternative explanation is that at the scale of these large regions, ammonia/ammonium deposition is a good surrogate measure for general leakiness of nitrogen from agriculture, and the factors that control export of nitrogen from agricultural sources to surface waters are strongly related in a linear fashion to the factors that regulate ammonia volatilization to the atmosphere; most ammonium deposition within a region comes from ammonia volatilization from agricultural sources within the same region since the transport of ammonia and ammonium through the atmosphere is limited (note that this assumption would not hold true for smaller watersheds). Thus, both ammonium deposition and the movement of nitrogen from agricultural sources into surface waters may be direct functions of the "surplus nitrogen" in agriculture (Howarth 1998). Ammonia/ammonium volatilization and deposition tends to be less than NO_y deposition in almost all regions of the North Atlantic Basin (Howarth et al. 1996b; see also table 2-2). Nonetheless, since ammonia/ammonium deposition may be a surrogate measure for agricultural nitrogen that moves directly into surface waters from sources such as leaching from feedlots and not just through volatilization to the atmosphere, the actual contribution of nitrogen from agriculture to the regional nitrogen exports could in theory be greater than that from NO_y deposition. Whether or not the pathway from agricultural sources to surface waters involves ammonia volatilization and redeposition, our result shows the importance of reducing the "surplus" nitrogen in agricultural and food systems (Bleken and Bakken 1997a, 1997b; van der Voet et al. 1996; Isermann and Isermann 1998).

Our large-scale regional analysis has proven a powerful tool for exploring controls on the movement of nitrogen from the landscape to estuaries and coastal seas. The mass-balance approach shows clearly that nonpoint sources of nitrogen dominate the movement of nitrogen to coastal waters

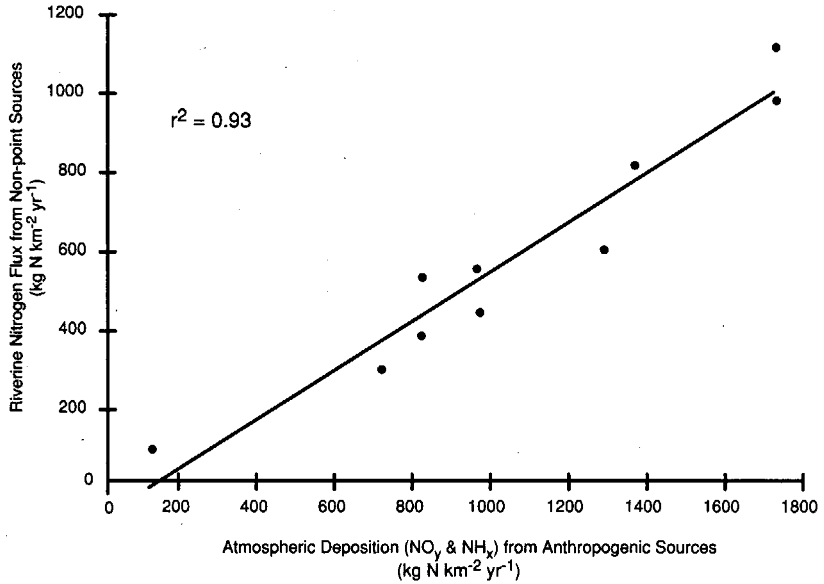

FIGURE 2-9 Riverine export of nitrogen from nonpoint sources as a function of the sum of wet and dry atmospheric deposition of oxidized and reduced forms of nitrogen from human-controlled sources. Reprinted from Howarth (1998) by permission.

and also indicate that only one-fifth of human-controlled inputs of nitrogen to regions reach estuaries and coastal seas. The fate of the majority of these human-controlled inputs of nitrogen to regions remains poorly known, although denitrification is likely to be the largest sink (Howarth et al. 1996b). Clearly, a critical research topic is to gain a better understanding of the fate of this "missing" nitrogen in the landscape. The correlational approach with the regional data needs to be interpreted with great care, as correlation cannot prove causation. Nonetheless, this approach suggests the great importance of fossil fuel combustion and subsequent deposition of NO_y onto the landscape as a major contributor to nitrogen inputs to coastal waters. The regression approach also suggests that the deposition of ammonia/ammonium may be a good surrogate, at the coarse scale of large regions, for nitrogen leakage from agricultural systems. Despite the power of working at large spatial scales for understanding nitrogen fluxes, much detail is lost. A better understanding of the controls on nitrogen flow through the landscape to surface waters probably requires inclusion of biogeochemical processes. We suggest that linking the regional approach to linked hydrological-biogeochemical models as we discussed for the Hudson River research may be a good approach to pursue next.

Acknowledgments

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CHAPTER 10

Gulf of Mexico Biological System Responses to Nutrient Changes in the Mississippi River

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Abstract

Freshwater inflow from the Mississippi River system is a major feature of the Louisiana continental shelf. This freshwater flow correlates closely with nutrient flux, surface-water net productivity, and bottom-water oxygen deficiency. Thus, changes in riverine inputs to the coastal ocean will likely affect biological systems on the adjacent shelf. Some changes are already happening as Mississippi River nutrient concentrations and loadings to the Gulf of Mexico have dramatically accelerated. Since the 1950s, the concentrations of dissolved N and P have doubled and Si has decreased by 50%, the dissolved Si:N ratio dropped from 4:1 to 1:1, and seasonal trends have changed. The resulting nutrient composition in the receiving coastal waters, on the average, shifted towards stoichiometric nutrient ratios closer to the Redfield ratio and more balanced than previously. Now, N and P are less limiting for phytoplankton growth, while some increase in Si limitation is probable. In spite of a probable decrease in coastal Si availability, the overall productivity of diatoms appears to have increased as evidenced by (1) equal or greater net phytoplankton community uptake of silicate in the mixing zone, compared to the 1950s; and (2) greater accumulation rates of biologically bound silica (BSi) in sediments beneath the plume. The increased percentage of BSi in Mississippi River bight sediments that parallels increased N loading to the system is direct evidence for the effects of eutrophication on the shelf adjacent to the Mississippi River. Composition shifts of individual phytoplankton species (heavily silicified diatoms → lightly silicified diatoms; diatom → non-diatom) indicates some responses to reduced Si supplies or changes in nutrient ratios or both. Finally, analysis of benthic foraminifera indicates an increase in oxygen-deficiency stress this century, with a dramatic increase since the 1950s. Increased bottom-water hypoxia could result from

increased organic loading to the seabed or shifts in material flux (quantity and quality) to the lower water column or both.

Introduction

The eutrophication of estuaries and enclosed coastal seas has increased over the last several decades, particularly in river-dominated ecosystems. Other evidence suggests a long-term increase in frequency of phytoplankton blooms, including noxious forms, for example, in the Baltic Sea, Kattegat, Skagerrak, and Dutch Wadden Sea (Smayda 1990). Also, an increase in the areal extent or severity of hypoxia or both was observed, for example, in Chesapeake Bay (Officer et al. 1984), the northern Adriatic Sea (Justić et al. 1987), and some areas of the Baltic Sea (see Andersson and Rydberg 1988). Hypoxia is also present in the northern Gulf of Mexico (figure 10-1), which receives the outflows of the Mississippi and Atchafalaya Rivers. In this presentation, we investigate the biotic responses of this ecosystem to changes in the delivery of nutrients and sediments by the Mississippi River.

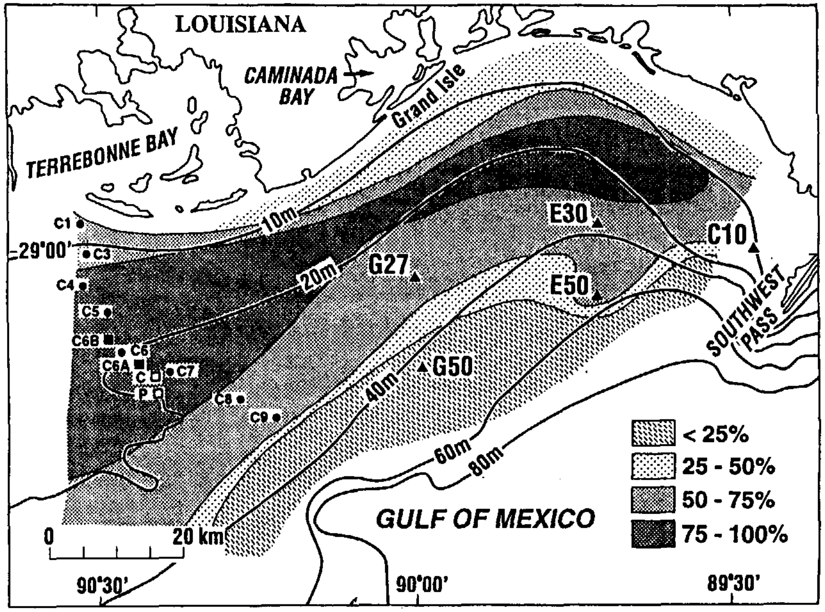

FIGURE 10-1 Location of study area within the Mississippi River Bight and off Terrebonne Bay for hypoxia monitoring; stippled intensity corresponds to frequency of occurrence of midsummer hypoxia at monitoring stations (1985–1987, 1990–1993, N. N. Rabalais, R. E. Turner, and W. J. Wiseman Jr., unpublished data). Transect C stations (closed circles), mooring locations, C6A and C6B (closed squares), LaSER stations (closed triangles), and "platform" and "control" stations of Fucik (1974) and Ward et al. (1979; open squares marked C and P).

The Mississippi River ranks among the world's top ten rivers in discharge and sediment yields to the coastal ocean (Milliman and Meade 1983). Freshwater inflow from the Mississippi and Atchafalaya (which carries one-third of the flow) is a major feature of the Louisiana shelf. Discharge from the Mississippi and Atchafalaya Rivers rapidly forms the Louisiana Coastal Current, a highly stratified coastal current that flows, on average, westward along the Louisiana coast and then southward along the Texas coast. Coastal winds are from the southeast at the beginning of the flood season. As spring progresses, the winds along the south Texas coast become favorable for upwelling, and the local flow reverses and returns back towards the north and east (Cochrane and Kelly 1986). As a result, density stratification, due primarily to salinity, persists for much of the year.

Fresh water attributed to the Mississippi and Atchafalaya Rivers may be traced as far west as Port Aransas on the Texas coast (Smith 1980) and 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). Within the northern Gulf of Mexico, the influence of the Mississippi and Atchafalaya outflows is evident in plots of surface salinity accumulated from numerous hydrographic cruises between 85°W (just east of Cape San Blas, Florida) to 95°W (just west of Galveston Bay, Texas) (figure 10-2). The influence of the freshwater input is more obvious to the west of the Mississippi delta (approximately 89°W) than to the east, with additional inputs near 91°W (the Atchafalaya delta). Plots of nutrient values (example given for silicate in figure 10-2) show a more pronounced decrease in concentration over an equal distance from the deltas when compared to the salinity plot. This indicates a nonconservative mixing due to biological uptake.

High biological productivity in the immediate (320 g C m^{-2} yr^{-1}) and extended plume (290 g C m^{-2} yr^{-1}) of the Mississippi River (Lohrenz et al. 1990; Sklar and Turner 1981; respectively) is mediated by high nutrient inputs and regeneration, and favorable light conditions. Small-scale and short-term variability in productivity are the consequence of various factors, such as nutrient concentrations, temperature, and salinity (Lohrenz et al. 1990, 1997), but on a seasonal time scale they are most influenced by Mississippi River flow and nutrient flux to the system (Justić et al. 1993). In this system, "new" nutrients become depleted along the river-to-ocean mixing gradient through dilution and biological uptake, and regenerated nutrients support primary production for great distances from the river mouth (Dortch et al. 1992a; Bode and Dortch 1996).

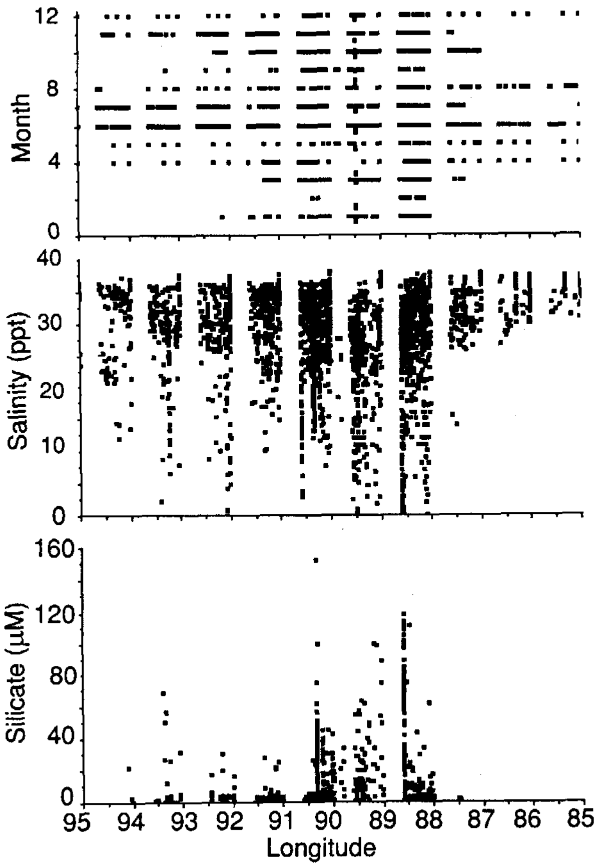

FIGURE 10-2 Plots of surface salinity and silicate concentration from a series of hydrographic cruises in 10–100 m water depth for 1972–1991, for months and longitudes indicated (modified from Rabalais et al. 1996).

Oxygen-depleted bottom waters are seasonally dominant features of the Louisiana continental shelf adjacent to the deltas of the Mississippi and Atchafalaya Rivers (Rabalais et al. 1991, 1992a, 1994a, 1994b, 1998). The areal extent of bottom-water hypoxia (< 2 mg l^{-1} dissolved oxygen) in midsummer may cover up to 20,000 km^2, with the spatial configuration varying interannually (figure 10-1). Samples along a transect on the southeastern shelf and continuous time series data off Terrebonne Bay document hypoxic bottom waters as early as February and as late as October, with widespread, persistent, and severe hypoxia or anoxia from mid-May to mid-September. Spatial and temporal variability in the distribution of hypoxia exists and is at least partially related to the amplitude and phasing of the Mississippi River discharge (Rabalais et al. 1994a; Wiseman et al. 1997). This oxygen deficiency is also related to the amplitude and phasing of nutrient flux to the coastal waters and subsequent production and flux of carbon from surface waters to the lower water column and seabed. The development and persistence of bottom-water hypoxia requires a strong, persistent pycnocline that prevents effective reoxygenation of deep waters from the upper layers. The fall destruction of the salinity-dominated pycnocline by wind mixing and thermal cooling of surface waters heralds the termination of hypoxic conditions.

The ability to detect changes in an estuary or coastal system is often hampered by the lack of data that are temporally and spatially complete enough to document adequately the variability of the system and deviations from the mean condition. This chapter draws on several syntheses of monitoring and hydrographic cruise data, comparisons with published data from prior decades, and historical records preserved in the sediments to document how changes in riverine nutrient fluxes have likely altered biological systems in the adjacent Gulf of Mexico. We present data from a variety of published syntheses that demonstrate in aggregate a close coupling between riverborne nutrients, net productivity, and hypoxia, as well as elucidate the effects of anthropogenic nutrient loads on a coastal marine ecosystem. First, we identify the links between riverine nutrient fluxes and biological response in a river-dominated coastal system. Second, we document long-term nutrient changes in riverine nitrogen, phosphorus, and silica fluxes to the continental shelf and probable changes in the receiving waters. We then address biological responses to these nutrient alterations, including shifts in probable nutrient limitations for phytoplankton growth, shifts in phytoplankton community structure, phytoplankton production, and indicators of eutrophication and increased oxygen stress. A similar presentation of results has been given previously in Rabalais et al. (1996).

Methods

While the influence of the discharge of the Mississippi and Atchafalaya Rivers can be measured at great distances from the deltas, we limit our discussion of consequences of riverine nutrient changes to the areas influenced by the immediate and extended plumes of the current birdfoot delta. In the Mississippi River bight these areas extend west to about 90°30′W or the entrance to Terrebonne Bay (figure 10-1). The oceanographic data synthesized include the following: (1) monitoring cruises conducted during the period 1985–1993 across the width of the Louisiana shelf, primarily in midsummer (stations covered by stippled areas in figure 10-1); (2) data from the southeastern shelf off Terrebonne Bay on a biweekly to monthly basis in 1985–1986 and 1990–1993 (transect C in figure 10-1); (3) more frequent sampling at an instrument mooring (stations C6A and C6B in figure 10-1); (4) six cruises covering 10 to 80 m water depth within the Mississippi River bight between 1987 and 1990 (that is, LaSER stations in figure 10-1); and (5) miscellaneous other cruises within the study area. Comparative data from the literature were available for periods in the 1950s and 1970s. Long-term river nutrient and flow data were available from the USGS and the U.S. Army Corps of Engineers. We also analyzed ^{210}Pb-dated sediment cores for historical biological records of overlying water productivity and oxygen stress. Research and statistical methods are provided in the primary literature as cited.

Synthesis Results

River-shelf Couplings

There is great daily and weekly variability in current flow and stratification on the shelf and, therefore, no simple description of the couplings between carbon production in surface waters and delivery and recycling in bottom waters at these time scales. However, there is evidence of an ecological "signal" (couplings) amid the "noise" (the variability) when more extensive data sets and longer time periods are examined.

Hydrographic data from the southeastern shelf in midsummer from 1985–1991 and off Terrebonne Bay on a biweekly to monthly basis in 1985–1986 and 1990–1993 (transect C in figure 10-1) were averaged by month and compared to long-term (1954–1988) Mississippi River flow at Tarbert Landing (Justić et al. 1993; figure 10-3). The surface layer (0 to 0.5 m) 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. The bottom layer (approximately 20 m), on the contrary, exhibits an oxygen deficit throughout the year with the maximum deficit 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 (Rabalais et al. 1991; Wiseman et al. 1997).

The correlation between Mississippi River flow and surface oxygen surplus peaks at a time lag of 1 month, and the highest correlation for bottom oxygen deficit is for a time lag of 2 months (Justić et al. 1993). 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 every N atom is recycled approximately four times, although recycling may be less important

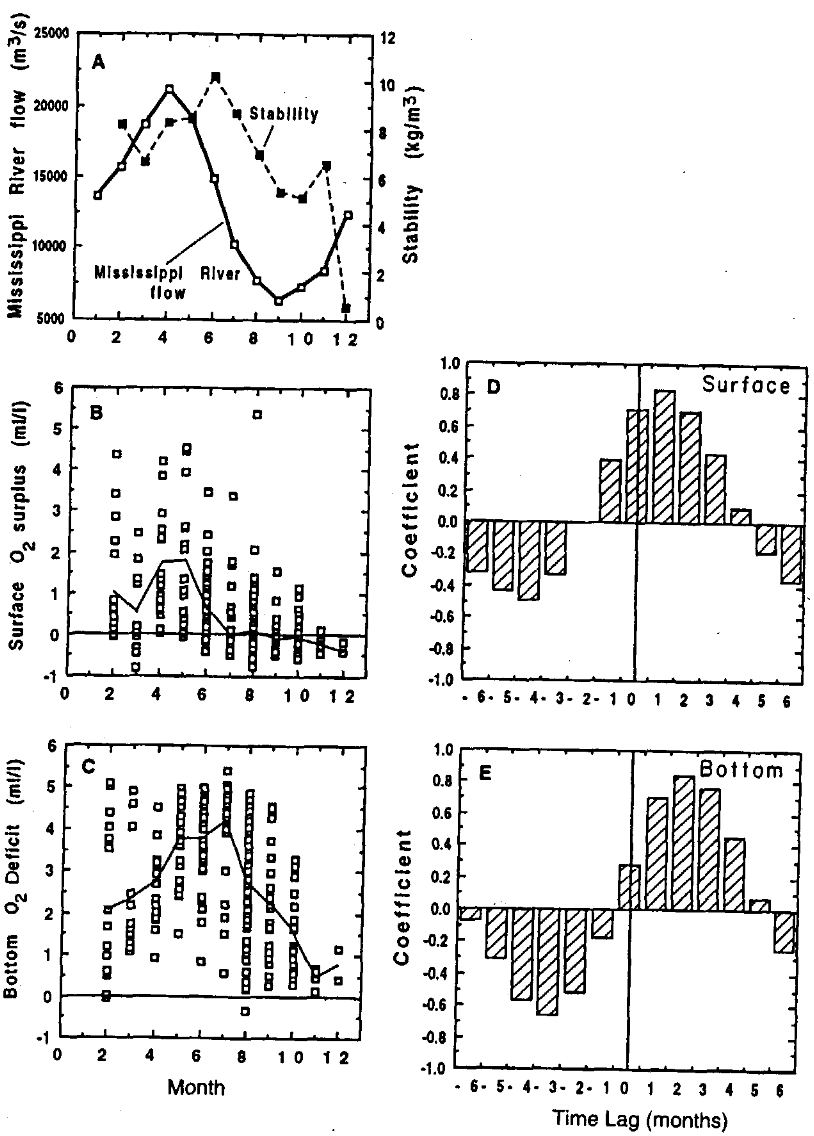

FIGURE 10-3 Monthly changes in Mississippi River flow and stability of the water column ($\delta\sigma_t$) (A), surface O_2 surplus (B), and bottom O_2 deficit (C) for stations within stippled area of figure 10-1 in 15 to 30 m water depth. The surface O_2 surplus and the bottom O_2 deficit were calculated as (O_2–O_2') and –(O_2–O_2'), respectively, where O_2 was the measured oxygen concentration and O_2' was the oxygen concentration at 100% saturation. Cross-correlation analysis of the data on river flow, surface O_2 surplus (D), and bottom O_2 deficit (E). Modified from Justić et al. 1995.

in the spring (Dortch et al. 1992a; Bode and Dortch 1996). This is an important finding, since 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 is an excess of organic matter derived from primary production that can be redistributed within the system; some of this will eventually reach the sediments. The development of summer hypoxia in the northern Gulf of Mexico (as exemplified by stations C6A and C6B, figure 10-1) is associated with the decay of organic matter accumulated during spring phytoplankton blooms (Qureshi 1995).

Changes in Lower Mississippi River Nutrients

Major alterations in the morphology of the main river channel, widespread landscape alterations in the watershed, and anthropogenic additions of nitrogen and phosphorus have resulted in dramatic water-quality changes this century (Turner and Rabalais 1991). Water-quality data for the lower Mississippi River was collected from stations at St. Francisville, Luling, New Orleans, and Venice (Turner and Rabalais 1991, 1994a; Rabalais et al. 1996). The mean annual concentration of nitrate was approximately the same in 1905–1906 and 1933–1934 as in the 1950s, but it has doubled in the last 40 years (results for St. Francisville for 1954–1994 shown in figure 10-4). The mean annual concentration of silicate was approximately the same in 1905–1906 as in the early 1950s, then it declined by 50%. Concentrations of nitrate and silicate appear to have stabilized, but trends are masked by increased variability in the 1980s and early 1990s data. We found no substantial records of total phosphorus concentrations in the lower Mississippi River before 1972. Although the concentration of total phosphorus appears to have increased since 1972, variations among years are large (Turner and Rabalais 1991).

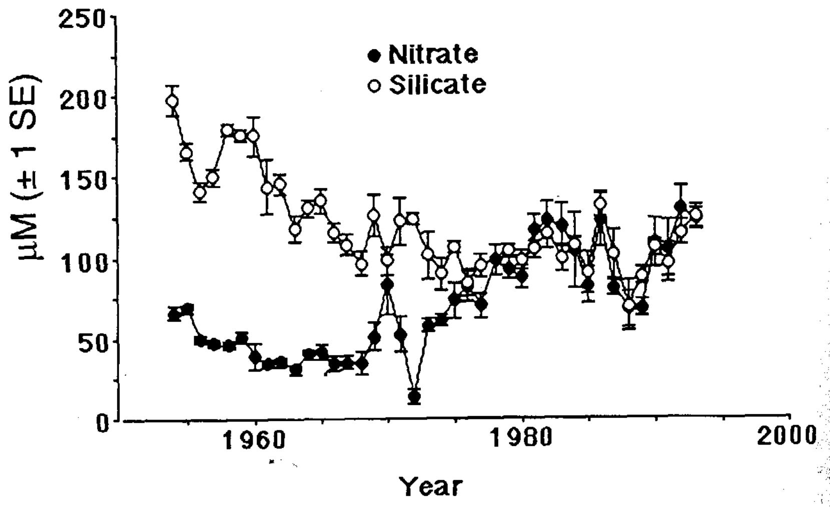

FIGURE 10-4 Average annual concentrations of nitrate and silicate from 1955 through 1994 in lower Mississippi River for the station at St. Francisville. Modified and updated from Turner and Rabalais (1991, 1994a); Rabalais et al. (1996).

The rise in nitrate since 1960 was coincidental with an increase in nitrogen fertilizer application in the Mississippi River watershed; fertilizer was applied in sufficient amounts to account for the changes in water quality (Turner and Rabalais 1991). The decrease in silicate since 1960 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 (see Schelske and Stoermer 1971; Schelske et al. 1986), thus reducing the annual supply of riverine silicate to coastal waters.

The silicate:nitrate ratios have also changed with the changes in nutrients (in Turner and Rabalais 1991, 1994a; Rabalais et al. 1996, and derivable from figure 10-4) from approximately 4:1 to 1:1. The ratio appears stable at 1:1 through 1994 with little variation. The seasonal patterns in nitrate and silicate concentration have also changed during this century (Turner and Rabalais 1991; Rabalais et al. 1996). There was no pronounced peak in nitrate concentration earlier this century, whereas there was a spring peak from 1975 to 1985, presumably related to seasonal agricultural activities. A seasonal summer-fall maximum in silicate concentration, in contrast, is no longer evident. The seasonal shifts in nutrient concentrations and ratios become increasingly relevant in light of the close temporal coupling of river flow to surface-water net productivity (1-month lag) and subsequent bottom-water oxygen deficiency (2-month lag) described above.

The proportions of dissolved Si, N, and P in the lower Mississippi River have changed historically and now closely approximate the Redfield ratio (Si:N:P = 16:16:1; Redfield 1958; Justić et al. 1995a, 1995b). We compared the data for two periods: 1960–1962 and 1981–1987 (table 10-1; data from Turner and Rabalais 1991). Substantial increases in N (300%) and P (200%) concentrations occurred over several decades, and Si decreased (50%). No data on total P concentration in the Mississippi River were reported prior to 1973; however, total P in the river showed a moderate increase between 1973 and 1987. By applying a linear least-squares regression on the 1973–1987 data, we estimated ($p < 0.01$) that the total P concentration increased twofold between 1960–1962 and 1981–1987. Accordingly, the Si:N ratio decreased from 4.2 to 0.9, the Si:P ratio decreased from 39.8 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 nutrient ratios for the Mississippi River (1981–1987 database) show an almost perfect coincidence with the Redfield ratio (Justić et al. 1995b). The proportions of Si, N, and P have changed over time in such a way that they now suggest a balanced nutrient composition.

Nutrient Changes on the Adjacent Continental Shelf

We 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 (Justić et al. 1994, 1995a, 1995b). Fully reliable long-term data sets to examine the nutrient composition 30 years ago, however, were not available. Accordingly, we reconstructed the past coastal nutrient composition (figure 10-5) by assuming that the relative proportion of nutrients in the river-dominated coastal waters reflect changing composition of riverine nutrients (table 10-1). This assumption takes into account that the Mississippi River is the most important nutrient source to the northern Gulf of Mexico. Also, a similar reconstruction technique for the northern Adriatic Sea produced results that closely paralleled the real data (Justić et al. 1995a, 1995b). The detailed reconstruction procedure is given in Justić et al. 1995a). By calculating the specific rates of change for Si:N, Si:P, and N:P ratios in the Mississippi River, we obtained a reasonable estimation of coastal nutrient composition 30 years ago. 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 the reconstructed nutrient ratios.

Comparison of measured and reconstructed nutrient ratios for the northern gulf adjacent to the Mississippi River outflow reveals long-term changes in proportions of nutrients in the surface waters (Justić et al. 1995a, 1995b; figure 10-5). The reconstructed nutrient ratios for 1960, on average, scatter further from the center of the grid (that is, the Redfield ratio) than the recent data. By applying the Redfield ratio as the criterion for balanced nutrient composition, P and N deficiencies have decreased while Si deficiency has increased. Equally important, recent nutrient ratios approximate the Redfield ratio, suggesting an almost perfectly balanced nutrient composition.

Probable nutrient limitation (Dortch and Whitledge 1992) was also assessed by comparing 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 (Justić et al. 1994) show that dissolved N concentrations in the surface layer of the northern Gulf of Mexico during the period 1985–1992 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

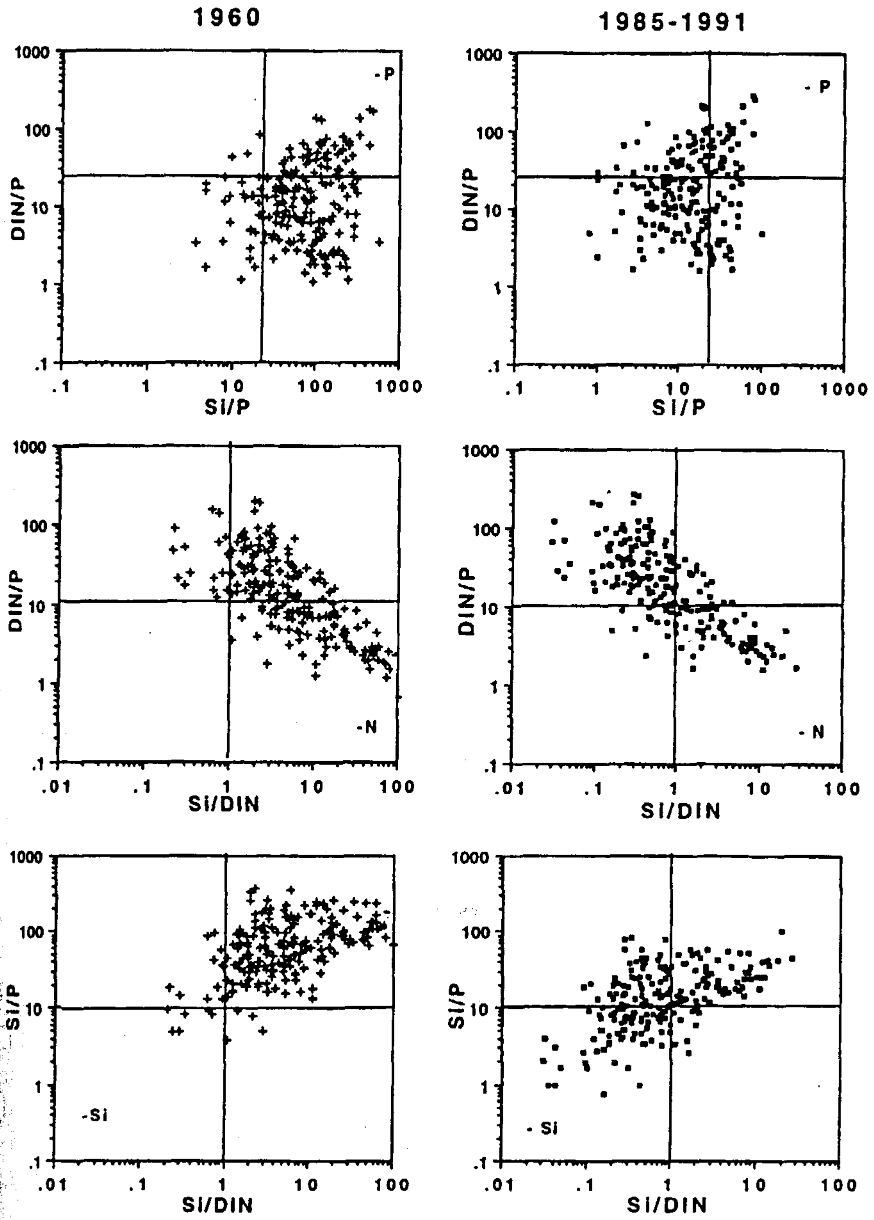

FIGURE 10-5 Scatter diagrams of atomic ratios of dissolved inorganic nitrogen (DIN), reactive phosphorus (P), and reactive silica (Si) in surface water of the northern Gulf of Mexico for (left) reconstructed data for 1960 and (right) ambient nutrient ratios during 1985–1991. From Justić et al. 1995a.

TABLE 10-1

Concentrations (μM) and atomic ratios of nitrogen (N), phosphorus (P), and silica (Si) in the lower Mississippi River and the northern Gulf of Mexico for 1960–1962 and 1981–1987; x = mean value, n = number of data, S = standard error, $p < 0.001$ = highly significant difference in nutrient concentrations between the two periods, based on a two-sample t-test. (Modified from Justić et al. 1995a, presented in Rabalais et al. 1996.)

		Mississippi River		Northern Gulf of Mexico	
		1960–1962	1981–1987	1960[d]	1985–1991
Nutrient concentration (μM):					
N[a]	x	36.5	114	2.23	8.13
	n	72	200	219	219
	S	2.9	6.0	0.16	0.60
		($p < 0.001$)			
P[b]	x	3.9	7.7	0.14	0.34
	n	—	234	231	231
	S	—	0.4	0.01	0.02
		($p < 0.001$)			
Si[c]	x	155.1	108	8.97	5.34
	n	72	71	235	235
	S	7.5	4.3	0.55	0.33
		($p < 0.001$)			
Average atomic ratios:					
Si:N		4.2	0.9	4.0	0.7
N:P		9	15	16	24
Si:P		39.8	14	64	16

[a] N-NO_3 for the Mississippi River, dissolved inorganic nitrogen (DIN = $NO_3 + NH_4 + NO_2$) for the northern Gulf of Mexico.

[b] Total P for the Mississippi River, reactive P for the northern Gulf of Mexico.

[c] Reactive Si.

[d] Reconstructed data.

were 39%, 41%, and 10%, respectively, in 1960. These findings are important because studies of nutrient uptake kinetics (Rhee 1973; Harrison 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 nutrient limitation has decreased.

Changes in Phytoplankton Species Composition

The changes in riverine and coastal nutrient concentrations and ratios over time should lead to observable changes in phytoplankton species composition. Published reports of phytoplankton species composition for 1955–1957 near

the delta (Simmons and Thomas 1962) and for 1972–1973 approximately 80 km west of the delta (Fucik 1974; Ward et al. 1979) were compared with recently obtained data (1990–1993) from near the delta and at stations C6A and C6B in 20 m water depth off Terrebonne Bay (figure 10-1; Dortch et al. unpubl. data; presented in Rabalais et al. 1996; tables 10-2 and 10-3). Dominant species for each area and time were picked by ranking the phytoplankton by concentration, summing the ranks, and picking the top-ranked groups. This comparison is qualitative because of differences in locations, seasons sampled, and methodology. The methodology used in earlier reports may have missed common non-diatoms, such as small coccoid cyanobacteria and phytoflagellates, which 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 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 non-diatom species composition from the 1950s and 1970s to present. Some heavily silicified diatom species are either not observed at all in recent samples (boldface in tables 10-2 and 10-3) or are much less dominant. For example, no *Melosira* species, which appear heavily silicified, were observed in 1990–1993, but these species were present in both 1955–1957 and 1972–1973. *Asterionella japonica* (=*Asterionellopsis glacialis*, Round et al. 1990) was observed recently at low salinities in the spring but is not a dominant species as it was from 1955–1957.

Similarly, more lightly silicified diatoms are documented for the 1970s and are present now, especially at higher salinities. *Rhizosolenia fragilissima* and *Ceratulina pelagica*, which are so lightly silicified they are sometimes difficult to see, were not reported for 1955–1957. Two species of *Leptocylindricus* spp. were frequently dominant in 1990–1993 but were a minor constituent in 1955–1957. During 1972–1973 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 the shift in dominant diatom composition toward less silicified species occurred between 1955–1957 and 1972–1973, but methodological differences preclude conclusions about changes in non-diatoms. Since Si availability continued to decrease after the early 1970s (Turner and Rabalais 1991), a continued shift in species composition to non-diatoms would be expected. The phytoplankton at C6A and C6B in 1990–1993 were often numerically dominated by small flagellates and cyanobacteria (not shown in

TABLE 10-2

Comparison of dominant diatoms near the river mouth and in the plume between 1955–1957 (Simmons and Thomas 1962) and 1990–1992 (Dortch et al. unpublished data) for spring and summer. Species in bold indicate those previously observed but not currently found or much less dominant, or vice versa. Where taxonomic names have changed, the original published name is retained. (Modified from Rabalais et al. 1996.)

	Spring			Summer	
	Near-River	Plume		Near-River	Plume
May 1957[a]	***Melosira* sp.** ***Melosira distans*** *Cyclotella* sp. *Cyclotella meneghiniana* *Asterionella japonica* *Chaetoceros* spp.	*Chaetoceros* sp. *Chaetoceros vanheurkii* *Chaetoceros affinis* *Thalassiosira nitzschioides* *Asterionella japonica* *Nitzschia* sp. *Skeletonema costatum* ***Eunotia* spp.**	June 1955[a]	*Cyclotella* spp. ***Melosira* spp.** *Skeletonema costatum* *Chaetoceros affinis*	*Cyclotella* spp. *Skeletonema costatum* *Chaetoceros affinis*
May 1991[b]	*Cyclotella* spp. < 8µm ***Extubocellulus spinifer*?** *Thalassiosira/Cyclotella*	***Dactyliosolen fragilissimus*** *Pseudo-nitzschia* spp. *Leptocylindrus minimus*	July 1990[b]	*Skeletonema costatum* *Cyclotella* spp. < & > 8µm *Thalassiosira* spp. *Thalassiosira/Cyclotella* Unident. centric > 8µm *Thalassionema nitzschioides*	*Skeletonema costatum* *Cyclotella* spp. < & > 8µm ***Dactyliosolen fragilissimus*** ***Ceratulina pelagica*** *Thalassiosira/Cyclotella* *Thalassionema nitzschioides*
			July 1991[b]		*Cyclotella* spp. > 8µm *Chaetoceros* spp. ***Ceratulina* sp. (not *pelagica*)** ***Dactyliosolen fragilissimus***
			July 1992[b]	*Thalassiosira/Cyclotella* *Skeletonema costatum* *Cyclotella* spp. < 8µm Unident. pennate < 8µm ***Dactyliosolen fragilissimus***	Sigmoid nitzschiod pennate < 8µm *Skeletonema costatum* *Cyclotella* spp. < 8µm

[a] Data for 1955–1957 are for four areas east of the Mississippi River delta (two with salinity < 18 ppt combined under "Near-River" and two with salinity of 18–32 ppt combined under "Plume." Most of the 1990–1992 data are from west of the delta but are matched for salinity and distance from the river (no stations west of 89.92°W). Methods used by Simmons and Thomas (1962) were reliable for diatoms > 0.6 µm.

[b] Water samples for 1990–1992 were preserved with glutaraldehyde and filtered onto 0.2, 3, and 8 µm polycarbonate filters (Dortch et al. 1992b). Organisms on the 3 and 8 µm filters were stained with proflavin. The 0.2 and 3 µm filters were counted immediately by epifluorescence microscopy; the 8 µm filters were frozen and counted later.

TABLE 10-3

Comparison of dominant phytoplankton species mid-shelf along the Louisiana coast during the periods 1972–1973[a] and 1990–1993.[b] High river flow in spring 1990 and 1993 make these periods most comparable. Species in bold indicate those previously observed but not currently or much less dominant or vice versa. Where taxonomic names have changed, the original published name is retained. (Modified from Rabalais et al. 1996.)

Year	Location	September	July
1972/1973	Platform	*Skeletonema costatum*	*Cyclotella striata* *Thalassionema nitzschioides* *Ceratulina pelagica* ***Melosira* sp.** *Coscinodiscus radiatus*
1972/1973	Control	*Rhizosolenia deliculata* *Gonyaulax monilata* *Gonyaulax turbynei* *Chaetoceros pelagicus* *Skeletonema costatum*	*Cyclotella striata* ***Melosira* sp.** *Coscinodiscus radiatus* *Pleurosigma navicalaceum* *Rhizosolenia schrubsolei*
1990	C6A	*Pseudo-nitzschia* spp. *Rhizosolenia delicatula* *Ceratulina pelagica* *Dactyliosolen fragilissimus* *Chaetoceros* spp.	*Dactliosolen fragilissimus* *Leptocylindrus minimus* *Pseudo-nitzschia* spp. *Cylindrotheca closterium*
1991	C6B	*Pseudo-nitzschia* spp. *Cyclotella spp.* < 8µm *Chaetoceros* spp. Unident. pennate < 8µm Nitzschioid-linear pennate < 8µm	*Cyclotella* spp. < 8µm *Cyclotella* spp. > 8µm Unident. auto.dino > 8µm *Thalassiosira/Cyclotella* *Scrippsiella* sp.
1992	C6B	*Chaetoceros curvisetus* *Chaetoceros socialis* *Chaetoceros* spp. *Cyclotella* spp. < 8µm *Ceratulina pelagica*	***Trichodesmium* sp.** *Leptocylindrus minimus* *Pseudo-nitzschia* spp.
1993	C6B	*Pseudo-nitzschia* spp. *Chaetoceros* spp. *Skeletonema costatum* Unident. centric > 8µm *Dactyliosolen fragilissimus*	*Chaetoceros* spp. *Thalassionema nitzschioides* ***Dinophysis caudata*** *Thalassiosera/Cyclotella* ***Ceratium furca***

[a] The data of Fucik (1974) and Ward et al. (1979) were obtained from "Control" and "Platform" (fig. 10-1). Their data indicate they used the standard Utermohl method on net tow samples; consequently, this table includes only diatoms, dinoflagellates, and large filamentous cyanobacteria, which they would surely have observed if they had been present.

[b] Data for 1990–1993 (Dortch et al. unpublished data) from nearby station C6A and C6B (fig. 10-1), methods in table 10-2.

table 10-3). They were not considered in this comparison, because it is not clear whether they would have been observed in the 1972–1973 study, even if present in large numbers. It is also tempting to hypothesize that the presence of *Trichodesmium* sp. in 1990–1993, but not in the earlier studies, indicates decreased Si availability. *Trichodesmium* sp., however, is a bloom-forming species, which has been previously reported for this region (Eleuterius et al. 1981).

Several species with importance to human health are now present but were either absent before 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 × 10^6 cells l^{-1} (Dortch et al. 1997). Several species known to produce domoic acid, which causes amnesic shellfish poisoning, have been identified in samples from Louisiana waters (Parsons et al. 1998) and high cellular domoic acid concentrations have recently been measured (Dortch, Doucette, Parsons, pers. comm.) Since 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 it could be responding to increasing N availability. *Dinophysis caudata*, a dinoflagellate that may be associated with diarrhetic shellfish poisoning (Dickey et al. 1992), was not recorded in the earlier studies. Recently, it was often present at low to moderate concentrations but was sometimes dominant and reached concentrations as high as 1 × 10^5 cells l^{-1}.

The increasing N availability and decreasing Si:N ratios appear to have led to increases in dominance of lightly silicified diatoms and non-diatoms. The indicated shift from heavily to lightly silicified forms may have altered carbon flux via directly sinking phytoplankton cells, if silica has become limiting. Subsequently, organic loading to the seabed and, possibly, oxygen depletion may have been affected. On the other hand, an increase in non-diatom forms that are less preferentially grazed than diatoms may have increased the flux of these cells into the lower water column and altered oxygen consumption in a different manner.

Silicate-based Phytoplankton Community Response

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 (figure 10-6). Uptake can be statistically modeled as a deviation from this mixing line, which we did for thirty-one adequately sampled data sets (Turner and Rabalais 1994a). We 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). We normalized for the effects of varying concentrations in the riverine end-member (see Loder and Reichard 1981) and compared the estimated net silicate uptake at 30 ppt as a function of silicate riverine end-member concentration (figure 10-6). Nonconservative 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 1979 than before 1979 (figure 10-6). The results from this analysis suggests that net silicate uptake in the dilution gradient from river to sea has remained the same, or even increased, as the riverine concentration of silicate decreased.

Biologically Bound Silica and Carbon Accumulation

As noted earlier, nearly coincidental trends in silicate (decreasing) and nitrogen (increasing) concentrations in the Mississippi River caused the average annual dissolved silicate:nitrate-N atomic ratio (Si:N) to decline from about 4:1 earlier this century to 1:1 this decade. The present Si:N ratio (1:1) is thought to be a significant limiting threshold for diatom growth, intraspecific competition, and production (Officer and Ryther 1980; Smayda 1990;

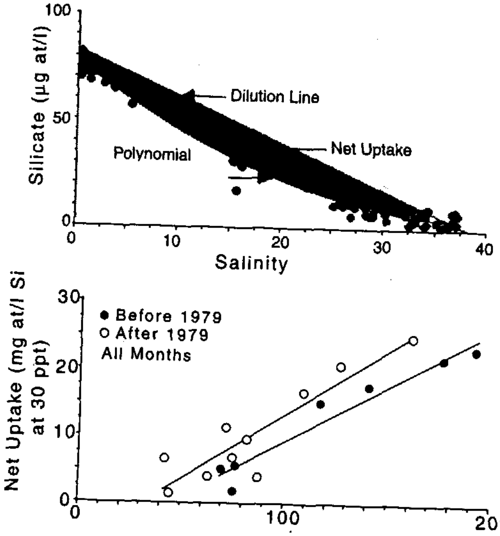

FIGURE 10-6 An example of the dilution of silicate in the Mississippi River (for June 1953) upon mixing with sea water (upper panel). The shaded area is the net up-ake of silicate not due to dilution. A polynomial line of best fit was used to estimate he silicate concentration at 20 and 30 ppt. The estimated net uptake of silicate above ilution losses at 30 ppt, for all months of data (lower panel). Modified from Turner nd Rabalais (1994a).

Dortch and Whitledge 1992; Turner and Rabalais 1994a). Thus, two contrasting hypotheses predict changes in phytoplankton on this coast since the 1950s. The first is that the coastal phytoplankton are nitrogen, not silica, limited, and higher nitrogen loading will result in proportionally higher phytoplankton production rates. A competing hypothesis is that the combination of lower silica fluxes and a Si:N ratio near 1:1 will result in lower production rates through limits on diatom production along with species-composition shifts. This is a nontrivial issue for managers, because diatoms are an ecologically important constituent of phytoplankton and contribute significantly to the organic loading of bottom waters and sediments and the subsequent oxygen depletion.

We documented that surficial sediments, directly downstream and beneath the surface riverine 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. 1992b; Turner and Rabalais 1994b). We further quantified the silica in the skeletal remains of diatoms sequestered as biologically bound silica (BSi) in dated sediment cores from the same region. The highest concentrations of BSi were in sediments deposited in 25 to 50 m water depth in the middle of the sampling area. The percent BSi in sediments from deeper waters (110 and 200 m) were generally stable through time but rose in the shallower stations (10 and 20 m) around the beginning of this century. At the intermediate depths (27 to 50 m), where both the percent BSi concentration and accumulation rates were highest, coincidental changes in the percent BSi with time were evident, especially in the 1955 to 1965 period (a rise and fall) and a post-1975 (1980?) rise that was sustained to the sampling date (1989; Turner and Rabalais 1984b; figure 10-7). The general pattern that emerges is a constant accumulation of BSi from 1800 to 1900, then a slow rise, followed by a more dramatic rise in the past two decades. Diagenesis of the BSi undoubtedly occurs in these cores but will be relatively low because the sedimentation rate is high (> 0.5 cm yr^{-1}). 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 produced by nutrient-enhanced growth, the more BSi will be deposited. Additional information is in Turner and Rabalais (1994b).

If the assumption is made that the BSi:C ratio at the time of deposition remained constant during this century, then the increased BSi deposition represents a significant change in carbon deposition rates (up to 43% higher in core sediments dated after 1980 than those dated between 1900 and 1960). The pattern in percentage of BSi changes parallels the documented increases in nitrogen loading in the lower Mississippi River, over the same period during which the silicate concentrations have been decreasing (figure 10-7). We conclude from our analyses that the flux of diatoms from surface to bottom waters, beneath the Mississippi River plume, increased this century. These changes were coincident with changes in riverine nitrogen loadings and resulted in higher organic sedimentation to bottom-water layers. The depletion of bottom-water oxygen, its persistence and areal coverage on this shelf, is thus indicated to have been altered this century.

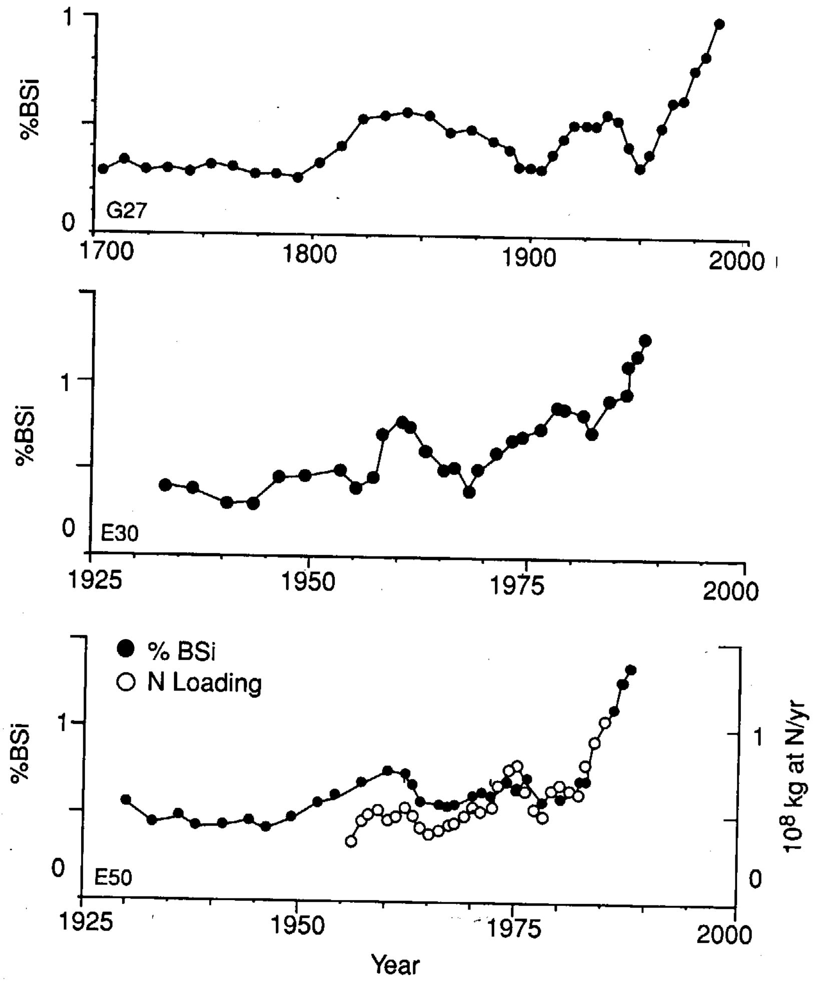

FIGURE 10-7 Average percentage of biologically bound silica (BSi) concentration of sediments in each section of ^{210}Pb-dated sediment cores from stations G27, E30, and E50. A 3-year running average for each sampling date is shown. The percentage of BSi data for station E50 are superimposed with a 3-year running average of the nitrogen loading from the Mississippi River through the delta passes. Modified from Turner and Rabalais (1994b).

Consequences to Hypoxic Bottom-water Formation and Severity

Long-term changes in the severity and extent of hypoxia cannot be assessed directly, because systematic sampling of bottom-water dissolved oxygen concentrations did not begin until 1985. Prior records of hypoxia, dating to 1973, were obtained sporadically as components of other studies; previous events were drawn from anecdotal relationships with shrimp trawl yields. Therefore, biological, mineral, or chemical indicators of hypoxia preserved in sediments, where accumulation rates record historical changes, provide clues to prior hydrographic and biological conditions.

Dominance trends of benthic foraminifera serve as indicators of reduced oxygen levels or carbon-enriched sediments or both (Sen Gupta et al. 1981; Sen Gupta and Machain-Castillo 1993). The same series of ^{210}Pb-dated sediment cores used for BSi analyses (Turner and Rabalais 1994b, figure 10-7) were used for determination of benthic foraminifera. Some downcore shifts in species abundance at station G27 (figure 10-7) in the Mississippi River bight were interpreted as foraminiferal responses to increasing oxygen stress (Sen Gupta et al. 1993, 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 indicates that the dominance of *Ammonia parkinsoniana* over *Elphidium* spp. is much more pronounced under hypoxia than in well-oxygenated waters. The relative abundance of *A. parkinsoniana* is also correlated with percentage of BSi (food source indicator) in sediments. In the context of modern hypoxia, species distribution in dated sediment cores reveals stratigraphic trends in the *Ammonia/Elphidium* ratio that indicate an overall increase in oxygen stress (in intensity or duration) in the last 100 years (figure 10-8). In particular, the stress seems especially severe since the 1950s. For this time period, both *Ammonia* and *Elphidium* become less important components of the assemblage, while *Buliminella morgani* shows an unusual dominance as also shown by Blackwelder et al. (1996). *Quinqueloculina* (a significant component of the modern assemblage only in well-oxygenated waters) has been absent from the record of this core since the 1870s but was a conspicuous element of the fauna in the previous 100 years. Thus, there are indications that oxygen-deficiency stress increased as nutrient loads and carbon flux to the seabed increased.

Predictions

One reason to synthesize data in estuarine and coastal studies is to understand processes and likely responses of the system to a variety of human impacts. Once a series of coherent patterns and interactions is outlined, as we have done for the continental shelf adjacent to the Mississippi River effluent,

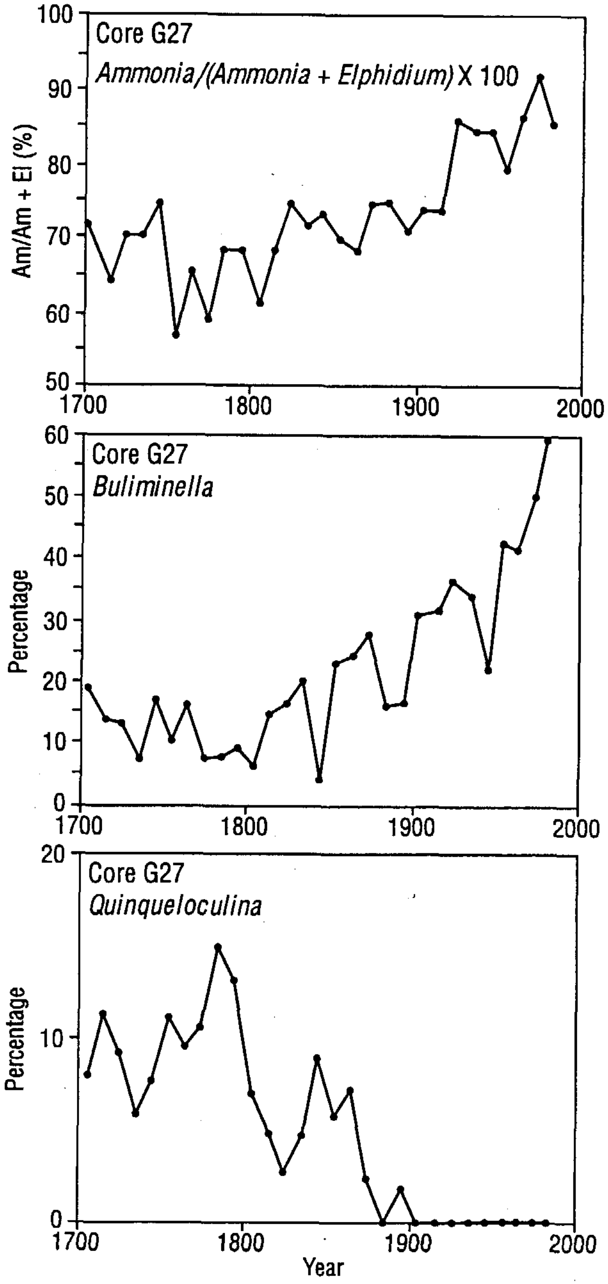

FIGURE 10-8 Changes in benthic foraminifera species with stratigraphic depth in ^{210}Pb-dated sediment core from station G27 in the Mississippi River Bight. From Rabalais et al. (1996).

it may be used to predict the response of the coastal system to alterations in riverine nutrient fluxes. Policies for nutrient control in freshwater systems seldom take into account the impact, or lack of impact, on coastal systems. Controls in freshwater systems often target phosphorus, based on the numerous laboratory and field studies of the stimulatory effect of phosphorus on freshwater ecosystems. Coastal systems, however, are usually thought to be nitrogen limited, at least part of the time, and this includes the shelf adjacent to the Mississippi River.

If freshwater phosphorus loading is reduced, we predict that the rate of BSi burial in freshwater sediments will decrease because freshwater diatom production will be lower (similar to the decreased percentage of BSi in more recent Lake Michigan sediments; shown in Conley et al. 1993). The release of dissolved Si from the sediments in the watershed will exceed the uptake of dissolved Si in the water column. The result will be an increase in dissolved Si within the Mississippi River and in the adjacent coastal waters (a return to the 1950s and turn-of-the-century levels). At the same time, controls on N use and loadings within the watershed may or may not be affected.

Based on the biological consequences of documented historical riverine changes, we foresee three possible scenarios for future system responses as riverine fluxes of silicate increase (figure 10-9). First, if N concentration remains the same, overall N limitations to phytoplankton productivity will be similar to present, but Si will no longer be limiting. The result would be increased BSi and carbon accumulation in the sediments, and an increase in the extent and severity of hypoxia. Second, if N concentrations increase, there will be no N or Si limitations and the Si:N ratio will be balanced. The result would be greatly increased BSi and carbon accumulations as well as substantial increase in the severity of hypoxia. Third, if N concentrations are reduced (for example, to 1950s values), then N would again return to the limiting nutrient status; although Si would be in abundant supply, the system would be restricted by N supplies and hypoxia would decrease.

If water quality improves, the close coupling between riverine nutrient loading and phytoplankton production should enable reversal of the current effects of nutrient increases and nutrient ratio changes. However, the management of one nutrient alone (Si or N) may not be sufficient to reduce eutrophication to an acceptable level if the compensatory qualitative adaptations of species lead to new phytoplankton communities, including those with noxious or toxic species.

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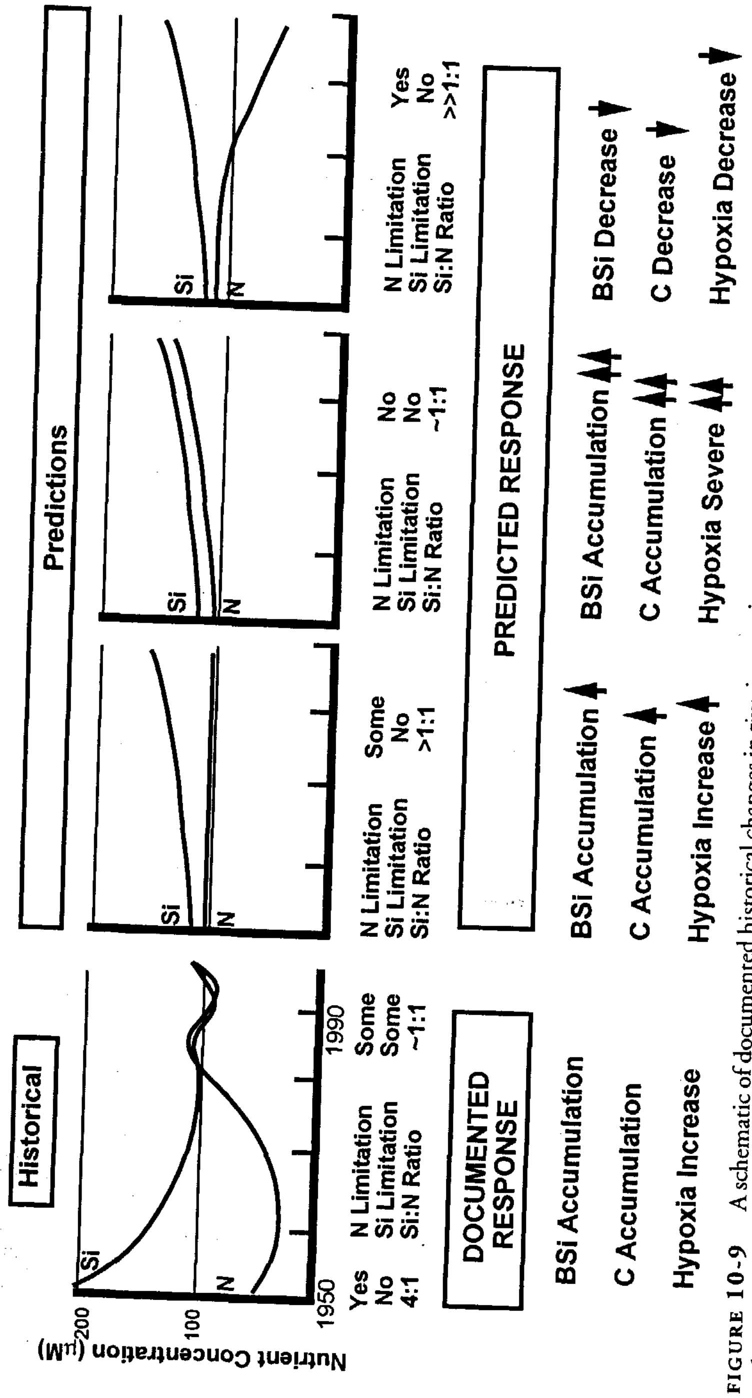

FIGURE 10-9 A schematic of documented historical changes in riverine nutrient concentrations, nutrient ratios, and biological responses, and a series of predicted responses depending on a constant increase of *silica* and varying changes in nitrogen loadings. A stronger response is indicated by double arrows. (See text for discussion; modified from Rabalais et al. 1996).

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CHAPTER 11

Influence of River Flow and Nutrient Loads on Selected Ecosystem Processes

A Synthesis of Chesapeake Bay Data

Walter R. Boynton and W. Michael Kemp

Abstract

In this chapter we assembled and analyzed two data sets, one a discontinuous 22-year time series (1972–1977, 1985–1993) of observations from a single mesohaline site in Chesapeake Bay, and the other, a much shorter time series from that site plus similar sites in four bay tributaries. For all locations, the data set includes measurements of river flow, nutrient-loading rate, phytoplankton primary production rates and biomass, water-column nutrient concentrations, and sediment-water exchanges of ammonium. In addition, data on sedimentation rates of chlorophyll *a* and bottom-water dissolved oxygen concentrations were analyzed at one site.

We examined a series of hypotheses concerning the influence of river flow and nutrient loading on these variables toward the goal of understanding underlying mechanisms. Significant relationships to flow and associated nutrient loads were found for all variables, some being stronger than others. In most cases, the influence of flow was found to extend over relatively short time periods (months to 2 years) and there were temporal lags between flow events and ecosystem responses on time scales of weeks to months. Results of analyses based on the time series from one location and on comparative analyses of data from five different sites were qualitatively similar; in this system it was not necessary to invoke comparative analyses to capture a large enough signal in forcing and response to observe interpretable patterns. Analyses generally indicated that relationships proximal to flow or nutrient loading rate were stronger (for example, nutrient load versus water-column nutrient mass) than those more removed from the direct influence of flow or nutrient load (for example, flow versus sediment nutrient releases).

These analyses indicate the importance of freshwater flow and associated nutrients in shaping chemical and biological responses in this estuary. Analyses are continuing and the next step will be to examine the effects of flow and nutrient loads on submersed vascular plant distributions and zooplankton and benthic communities.

Introduction

During the last decade there has been an increasing number of environmental measurements taken in coastal and estuarine systems, and this trend seems destined to continue for the foreseeable future. In part, this activity has been stimulated by increased awareness of natural resource deterioration in these environments due to human activities in the drainage basin as well as in the receiving water bodies. Common now are reports of declining or collapsed fisheries, toxic algal blooms, development of hypoxia and anoxia in deeper waters, and loss of submersed aquatic vegetation communities (Nixon 1990).

Despite much larger databases for many of these systems, we are still unable to confidently answer many fundamental questions concerning how these systems work and, from a practical viewpoint, what resource managers need to do to reverse declines in water quality and abundance of living resources. One reason for this state of affairs is that analyses and interpretations of these data sets have been limited. This is particularly true for many data sets collected in monitoring programs and ad hoc field surveys. In addition, scientific data collected in research programs are often interpreted within relatively narrow areas of scientific interest having little value at the larger scales of organization relevant for resource management (Malone et al. 1993). To be useful, these data need to be pulled together into some sort of synthesis that focuses on time, space, and organizational scales appropriate to the questions being asked.

In recent years, some very ambitious numerical efforts have been initiated and serve as one type of data synthesis. For example, sophisticated hydrodynamic models have been developed for a number of estuarine systems (for instance, Long Island Sound, Tampa Bay, Chesapeake Bay) and act as a framework for synthesis of large data sets as well as forecasting tools. In other cases, these models have been coupled with water-quality models and used as diagnostic tools in water-quality management programs, as is the case in Chesapeake Bay (Cerco and Cole 1992). While these tools have obvious advantages, they are expensive and time consuming to construct, analyze, and maintain.

The purpose of this chapter is to present the results of a direct, empirical type of synthesis whereby variations in key properties of coastal ecosystems are related to changes in riverine nutrient loading (Rigler 1982; Peters 1991). Specifically, we describe here the results of regression modeling based on data collected in Chesapeake Bay. In this work we have primarily focused on examining the influence of freshwater inputs (and associated nutrient loads) on several ecological processes. The importance of freshwater inputs is obvious; it is a central feature in the definition of estuarine systems, it influences physical dynamics (Boicourt 1992), is well correlated with nutrient inputs (Summers 1993), and has been implicated in regulating either directly or indirectly estuarine processes ranging from primary production (Boynton et al. 1982; Cloern et al. 1983) to benthic secondary production (Flint 1985) to fish recruitment (Stevens 1977) and catch (Sutcliffe 1973; Sutcliffe 1977; Ennis 1986). The emphasis here is the exploration of data sets for patterns that conform to expected relationships or suggest new relationships (see Meeuwig et al. 1998) rather than statistical testing for significant differences or temporal trends. We wish to examine environmental data for relationships and to use these as clues to suggest underlying mechanisms.

Approach and Methods

Conceptual Model

The focus of these analyses is to investigate the influence of river flow and associated nutrient inputs on selected ecological processes in Chesapeake Bay. Most, if not all, of these hypothesized direct or indirect effects of river flow on ecological processes have been documented in other systems. For example, phytoplankton biomass and community composition have been shown to be regulated by river discharge in San Francisco Bay (Cloern et al. 1983) and Texas estuaries (Flint 1985), while buoyancy effects of fresh water have been extensively investigated in various estuaries (Boicourt 1992), and responses of benthic respiration and nutrient regeneration to variations in phytoplankton production and deposition have also been examined (Flint 1985; Cowan et al. 1996). Here we consider the extent to which these effects of river flow are manifest in Chesapeake Bay and we have organized this analysis around a simple conceptual model (figure 11-1). In this model, river flow adds directly to the nutrient pools (1) and influences buoyancy of the water-column. River flow also determines the geographic positioning of water-column events (that is, events such as plankton blooms tend to shift seaward in high-flow periods and landward in low-flow periods) and the location of water-column deposition of organic matter to the benthos. Phytoplankton production (2) and biomass (3) are responsive to nutrient pools and phytoplankton biomass is lost to the benthic community via sinking (4). The benthic community recycles nutrients to the water column (6). Finally, deep-water dissolved-oxygen depletion (5) is influenced by stratification of the water column, organic matter derived from

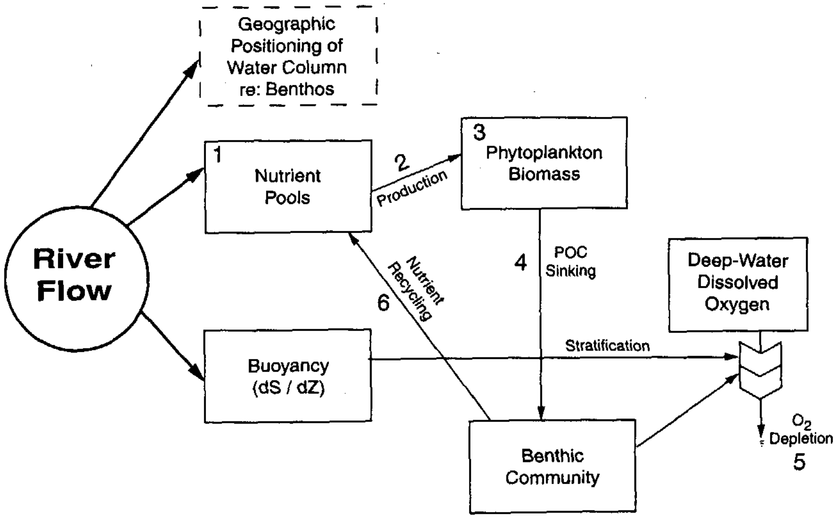

FIGURE 11-1 A simple schematic diagram showing the influences of river flow on ecosystem stocks and processes examined in this study. The mechanistic relationships between river flow and the stocks and processes shown in the diagram are explained in the text.

phytoplankton, and respiration of this organic matter by the benthic community.

Obviously, the perspective in this conceptual model is heavily biased toward bottom-up (as opposed to top-down) control of ecological interactions. We recognize that top-down effects can be important, and dominant, in some estuarine situations. For example, Alpine and Cloern (1992) found dramatic changes in the temporal pattern of phytoplankton production and biomass in San Francisco Bay following the introduction of a suspension-feeding clam. Meeuwig et al. (1998) found that herbivory by mussels was a strong modifier of algal biomass-nutrient relations in some Canadian estuarine systems. There are also more numerous and well-known examples from limnology (Carpenter and Kitchell 1988). In this preliminary analysis, we chose to emphasize the bottom-up perspective for simplicity and elegance.

Ecological Inference

Our approach to ecological synthesis consists of several steps, the first of which involves developing empirical models to specify the relationships of interest and to aid in selection of appropriate variables. Rigler (1982) differentiated ecological research into empirical and explanatory categories, the former with the goal of prediction and the latter with mechanistic explanations of predictions. In regression modeling the mechanistic relationships between variables are not specified while in explanatory approaches (such as simulation modeling) every interaction is exactly specified. In a sense, empirical approaches are more holistic while explanatory schemes are more reductionistic.

As an example of empirical model development, interannual variability in algal biomass may be of interest and appropriate variables might be chlorophyll *a* concentrations or some other measure of algal stock. The next step is to select a group of probable causative variables and in the above example these might be river flow, nutrient-supply rate, light availability, or others. Much of the above obviously requires previous knowledge in establishing relationship between variables so there is a natural interaction between reductionist and holistic approaches. We also recognize the need to establish alternative hypotheses in empirical approaches as suggested by Peters (1991). The idea here is to explore all reasonable explanations, rejecting most because they do not support the hypothesis and leaving us with one (or more) that can be supported and further explored. It is this step that largely differentiates this approach from a simple statistical examination of a data set for statistically significant relationships.

We have also employed comparative approaches in a portion of the analyses presented in this chapter wherein similar data from a variety of systems are used in the analysis. This technique has the advantage of increasing the signal range for both independent and dependent variables and hence increasing the chance of interpretable patterns emerging from what is admittedly a complex set of interactions (Vollenweider 1976; Nixon 1988). However, comparative approaches generally require "scaling" of variables in a fashion that makes them comparable among systems and this in itself can be a complex and interesting problem (Schnieder 1994).

In this chapter we present two groups of empirical analyses; the first is based on a data set collected at one location in Chesapeake Bay for a number of years (~13 years) while the second examines similar issues but uses a comparative approach based on data collected at multiple locations in Chesapeake Bay for shorter periods of time (1 to 4 years). We take advantage of a long data record in the former and inherent differences among systems in the latter; in a sense this can be thought of as a time-space substitution with both approaches being useful in testing ideas about ecosystem behavior (Pickett 1991).

Study Area

Chesapeake Bay is the largest estuary in the United States, having an area of 6,500 km^2, a length > 300 km, a width of 5–30 km, and mean depth of 8.4 m; it is closely embraced by the land (drainage basin surface area: bay surface area = 28:1). The surface area of the bay system is equally divided between the mainstem bay and the numerous (approximately thirty) tributary rivers and bays; however, about 66% of the volume is contained in the deeper mainstem (figure 11-2).

The hydraulic fill time (volume divided by freshwater inputs) is approximately 1 year and water residence times range from 3 to 6 months. The mainstem bay is stratified from late winter through early fall; stratification in tributaries is generally weaker and less persistent (Boicourt 1992). Water-column stratification is in part responsible for chronic hypoxic and anoxic conditions in deeper regions of the system (Kemp et al. 1992). It appears that the volume of hypoxic water has increased since colonial times (Cooper and Brush 1991), much of it in the last three to four decades (Boicourt 1992).

The bay and its watershed lie in the coastal corridor of dense human population between New York and Virginia; population in the watershed is now 13.6 million and is projected to soon be 16.2 million (Magnien et al. 1995). Current nitrogen- and phosphorus-loading rates averaged for the entire bay are about 13 gN m^{-2} yr^{-1} and 1 gP m^{-2} yr^{-1}, respectively; however, loading rates to distinct portions of the bay system range from both a factor of 5 higher and lower than these and thus provide a good opportunity for comparative analyses. Since European settlement, bay-wide loading rates of nitrogen and phosphorus have increased about six- and seventeenfold, respectively (Boynton et al. 1995).

One of the important characteristics of estuarine systems such as Chesapeake Bay is temporal variations in inputs such as freshwater flow (figure 11-3). During the past several decades, the magnitude of annual average freshwater input to the head of Chesapeake Bay has varied by a factor of 2.4; average annual flows from the Susquehanna River are about 1,200 m^3 sec^{-1} and represent about 50% of the freshwater flow to the entire Chesapeake system. However, seasonal patterns of flow are even more variable, especially during the "spring freshet." This important hydrological event has occurred between January and May in recent decades, though typically during March or April, and has varied in magnitude by a factor of 5 (figure 11-3).

Data Sources

Chesapeake Bay and associated tributaries is one of the most studied estuarine systems in the United States and a tremendous amount of data are available, especially from the last decade. Much of this information has been collected as part of the EPA's Chesapeake Bay Program and closely related monitoring programs conducted by the states of Maryland and Virginia.

To provide some indication of the intensity of this program, about 40,000 measurements of such variables as chlorophyll *a* have been made in the mainstem bay alone during the last decade. During the last decade the challenge

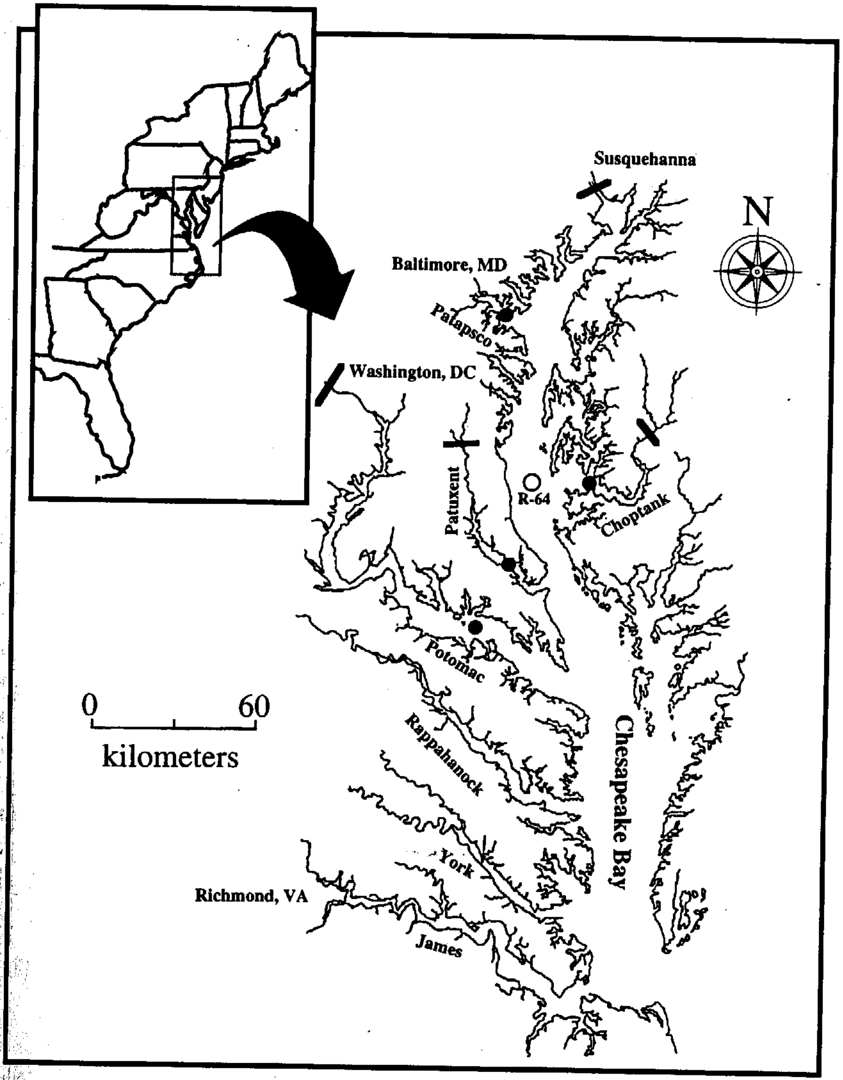

FIGURE 11-2 A map showing Chesapeake Bay and major tributary rivers and location of this estuarine system on the East Coast of the United States. Bold lines indicate locations at the fall line where freshwater inflows and nutrient loads are monitored. Bold dots indicate locations where water-quality, phytoplankton, and sediment-water flux measurements were made; these measurements as well as sedimentation rates were made at the site indicated by the open circle (R-64).

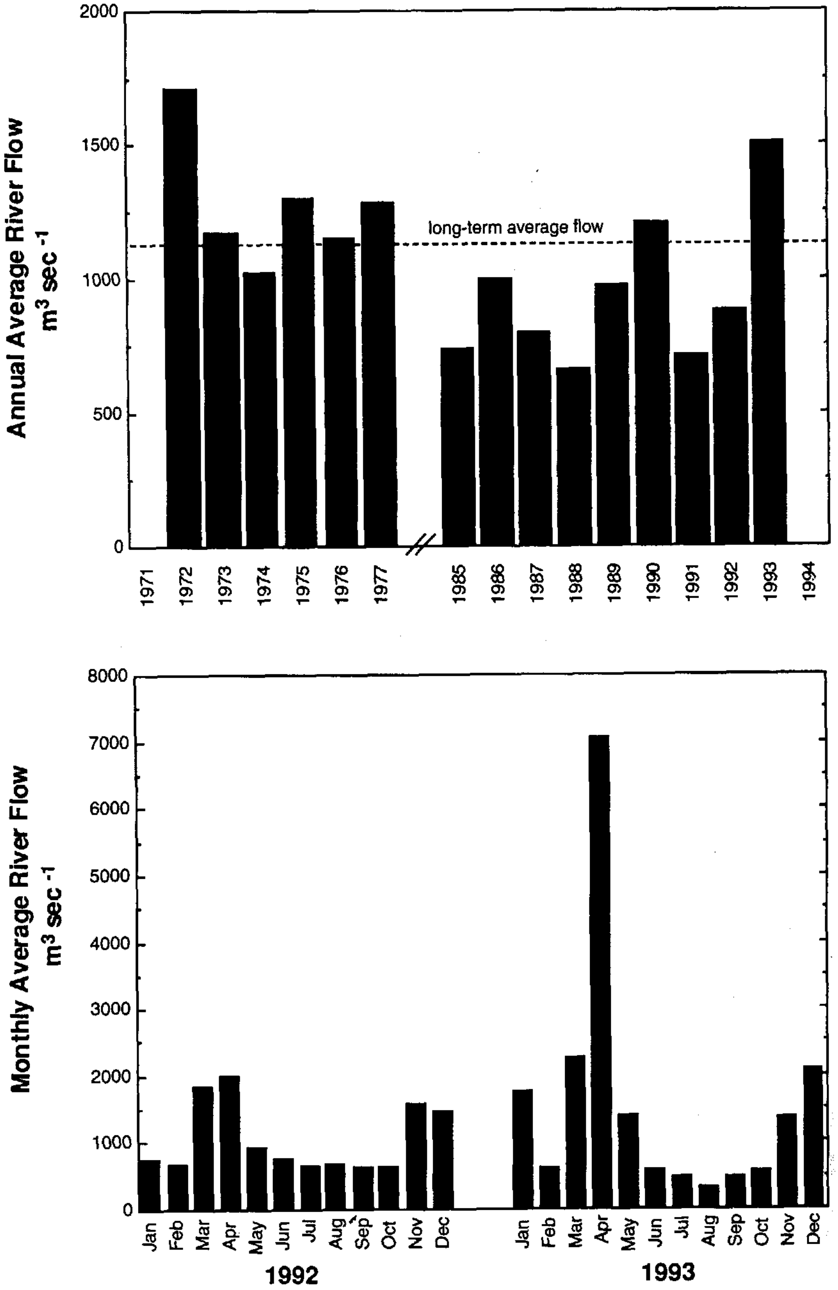

FIGURE 11-3 Estimates of annual average river flow (1972–1977 and 1985–1993) from the Susquehanna River entering the head of Chesapeake Bay. Flows are for the years used in regression analyses. Also shown are monthly average flows for the years 1992 and 1993 from the same location to indicate the variation in magnitude and seasonal patterns of freshwater inputs (USGS 1993).

of simply finding enough data to see if interpretable seasonal patterns exist has changed to one of managing large databases in such a way that any patterns present can be found. A brief description of the data collection program is provided in table 11-1 for the variables used in the analyses presented here; details concerning collection and analytical methodologies have been described in detail elsewhere (see sources listed in table 11-1).

Statistical Methods

Regression techniques have been applied to ecological problems for quite some time, often yielding useful results. In part, the attraction of this approach lies in its simplicity. As opposed to water quality, fisheries, and ecosystem models, data requirements can be relatively small, the time required to explore many possible relationships short, and there are few, if any, assumptions to be made concerning the form of mechanistic relationships. The relative ease of using the technique makes it, therefore, very attractive as a tool for scanning data in search of suspected relationships and as a tool that often suggests new interpretations.

While regression can be straightforward there are, of course, limitations that are both practical and conceptual. As with any statistical technique, strong correlation between variables does not, however tempting, indicate causal relationships. In addition, covariance among variables can lead to spurious conclusions. Assumptions concerning the distribution of data and other criteria for strict application of parametric techniques are often difficult or impossible to check.

In spite of these problems, and the apparent simplicity of the approach, there have been many useful applications of regression techniques to ecological problems. During the 1960s and early 1970s, Vollenweider and his colleagues developed a series of "mixed reactor regression models" relating algal standing stock (used as an indicator of trophic condition) to nutrient loading (primarily phosphorus loading to lakes). They found significant relationships that were useful in classifying lakes according to trophic status and in suggesting the amount of nutrient loading needed to change the trophic status of a lake (Vollenweider 1976). Similar, but less inclusive, efforts have been made relative to estuarine systems (Boynton et al. 1982; Meeuwig et al. 1998). It has long been taken as a fundamental tenet of ecology that there is some relationship, probably complex, between rates of primary production and fishery yields. Such a relationship was documented by Oglesby (1977) for lakes and later by Nixon (1988) for estuarine, coastal, and marine systems. More complex relationships between standing stock size, growth rates, and production in marine food webs have also been determined using regression modeling approaches (Sheldon et al. 1977; Ennis 1986).

TABLE 11-1

Brief description of data sources used in development of regression models presented in this chapter. Each program component is a part of the Chesapeake Bay Water Monitoring Program, which was initiated in 1984 and continues through the present time (Magnien et al. 1995). Phytoplanktonic production and chlorophyll *a* data from the 1972–1977 period are from Mihurshi et al. (1977).

Program Component	Variables Measured	Number of Stations	Field Technique	Sampling Frequency and Duration	Reference
Water-quality variables	T, S, DO, pH, chl *a*, dissolved and particulate N, P and Si concentration (vertical profiles)	50	One to five water column samples depending on total depth.Standard oceanographic analytical techniques.	16–20/yr 1984–present	Magnien et al. 1994
Freshwater and nutrient-loading rates	T, DO, conductivity, pH, chl *a*, *E. coli*, BOD, COD, TSS, total and dissolved N,	Fall line of all major rivers	Standard river gauges estimating daily flow.Regular parameter sampling and statistical modeling of flow-concentration relationships.	1–4/month (daily flow) 1972–present	Summers 1993
Sedimentation rates	Sedimentation rates of C, N, P, Si, chl *a*, and seston	One site in middle Chesapeake	One fixed vertical array. Collecting cups positioned in upper mixed layer, just beneath the pycnocline, and 1 m above the bottom.	Spr, sum, fall (~1/week) 1984–1993	Boynton et al. 199
Sediment-water exchange rates	Net sediment exchanges of O2, NO2, NO3, DIP, Si, CO2	8	Estimated from shipboard incubation of intact sediment cores. Incubations were under ambient conditions.	Spr, sum, fall 1/month 1984–1996	Boynton et al. 19
Phytoplankton component	Primary production rates, chl *a* concentration, and species composition	34	Short-term (3-hr), constant light ^{14}C incubations. Fluorometric chl *a* determinations.	16–20/year 1984–present	Sellner 1993

Results and Discussion

Time-series Observations at a Single Site

In this section, we examine the influence of river flow on phytoplanktonic production and biomass, deposition rate of spring-bloom phytoplankton, deep-water dissolved-oxygen declines, and recycling of ammonium from estuarine sediments. General pathways of the influence of river flow on these processes are summarized in figure 11-1. All of these analyses are based on time-series data collected at one location in the central portion of Chesapeake Bay (R-64, figure 11-2). We used seasonally or annually averaged data (6–8 or 16–20 observations, respectively) in these analyses rather than single, instantaneous values because these were the time scales of interest and because we wanted to avoid short-term variablity related to organism response times, changes in water residence times, and the like. These results were selected to serve as examples of the utility of synthesis as a framework to think about interrelationships of estuarine processes; there is a great deal of additional analysis that could be conducted on these and other data sets.

Algal Production and Biomass

The starting point for these investigations was suggested by previous analyses from lakes (for example, Vollenweider 1976) and coastal and estuarine systems (for example, Boynton et al. 1982; Nixon 1988) where statistically significant relationships were found between nutrient-loading rates and algal production and algal biomass. In our case, we used river flow as the independent variable because in Chesapeake Bay it is strongly correlated with nutrient-loading rates (Summers 1993) and provides most of the buoyancy that results in seasonal water-column stratification and hence definition of the upper mixed layer (Boicourt 1992).

We were initially doubtful about the possibility of finding strong relationships between flow and algal parameters. Previous investigators had adopted comparative approaches to obtain a sufficiently large range in loads and phytoplanktonic responses to observe significant relationships (Nixon 1988). Our initial concept was that there were so many factors controlling algal parameters that any one variable, even one like river flow that has multiple influences on the system, would explain only a small portion of the observed variability.

It appears that this is not the case. Results indicate strong relationships of river flow to biomass and, to a lesser extent, production (figure 11-4). In both, a large percentage (59% and 78%) of interannual variability was explained by river flow alone. This result reinforces the general conclusion that river flow is a dominant factor regulating some basic ecosystem processes in systems like Chesapeake Bay.

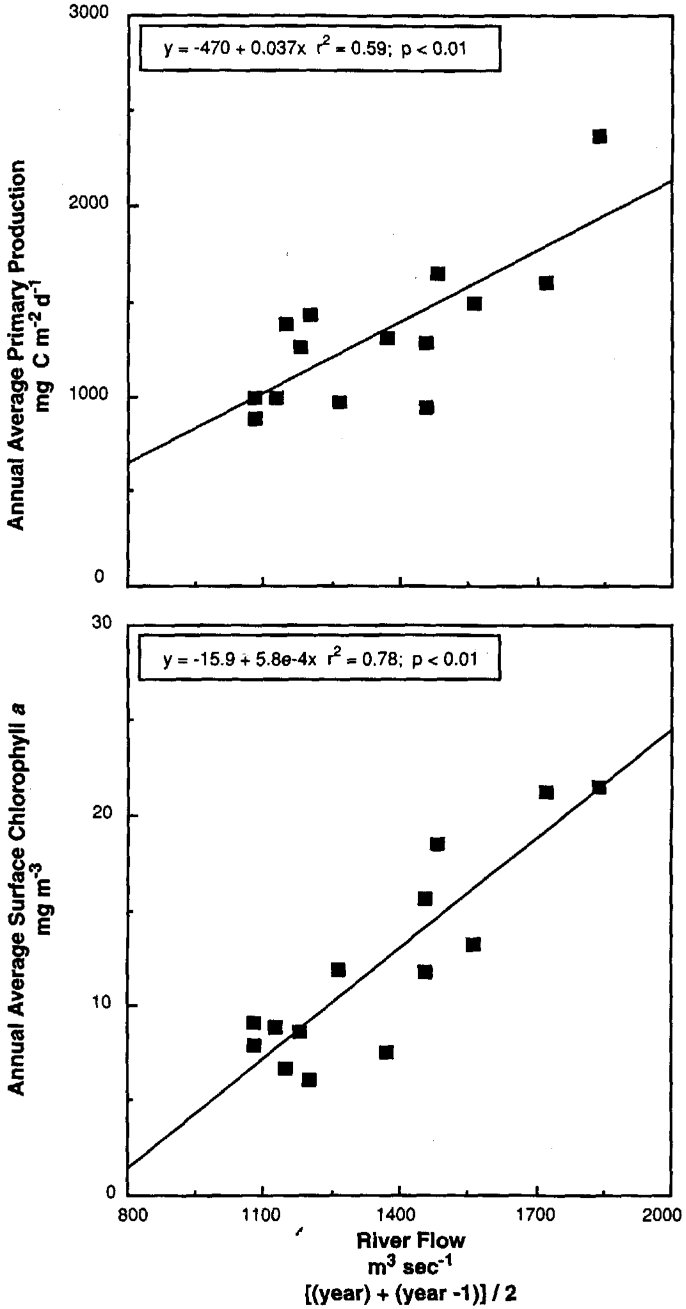

FIGURE 11-4 Results of linear regression models showing relationships between annual average phtoplankton primary production and surface chlorophyll *a* concentrations and freshwater flows from the Susquehanna River. River flow was calculated as the average of flow in the present and preceding year. Production and chlorophyll *a* data are from the R-64 site during the periods 1972–1977 and 1985–1993.

In this analysis several variations of independent (river flow) and dependent (phytoplankton production and biomass) variables were also tested, each being a variation representing a modified hypothesis concerning river influence on algal parameters (table 11-2). For example, in some trials just the spring freshet was used as the flow variable to explore the idea that this short but high period of flow was a key event in the annual or summer portion of the phytoplankton cycle. Many were statistically significant indicating the general importance of river flow. However, the flow variable that explained the most variability was an average of annual flow from the current year and from the previous year. This combination was suggested by examination of field data where it was noticed that production and biomass in years of average flow were higher than expected if they were preceded by a year of exceptionally high flow. This, in turn, suggests some nutrient retention or "nutrient memory" over time scales of a year rather than seasonal periods as suggested by bay water residence times (Boynton et al. 1990). Given the shallow depths of the bay, interannual retention of nutrients in the water column is not likely. The only likely multiyear nutrient storage site is sediments (Boynton et al. 1995). We suggest that in years of especially high flow, above-normal algal biomass is generated during the spring bloom. Recycling of this material supports high production through summer, which serves to conserve nutrients in the bay and make possible a large fall bloom. The deposition of the fall bloom to sediments, coupled with falling water temperatures, preserves nutrients through winter and they become available the next spring to support production and algal biomass at higher than expected levels. Kemp and Boynton (1984) proposed a similar sequence of

TABLE 11-2

A summary of results from linear regression analyses examining data sets for relationships between river flow and phytoplankton production and biomass. Entries in the table are r^2 values. Single and double asterisks indicate significance at the 0.05 and 0.01 probability levels, respectively. The number of observations in each analysis was fourteen. Data are from Mihurski et al. (1977), USGS (1993), and Magnien et al. (1994).

	River Flow Averages					
Phytoplankton Variables	Jan.–Mar.	Jan.–May	Jan.–Jun.	Jan.–Jul.	Avg. Annual	Two-Year Avg.
Annual average chlorophyll *a*	0.22	0.50**	0.67**	0.47**	0.64**	0.78**
Annual average production	0.14	0.20	0.15	0.13	0.22	0.58**
Summer average chlorophyll *a*	0.17	0.47**	0.74**	0.74**	0.62**	0.56**
Summer average production	0.18	0.31*	0.25	0.23	0.26	0.51**

events based on observations made in the Patuxent River estuary, but in that case the sequence did not include more than one annual cycle.

Spring-bloom Deposition

In most years, the annual cycle of phytoplankton biomass accumulation exhibits a distinct maxima associated with the spring bloom in the mesohaline regions of the Chesapeake Bay and there is considerable interannual variability in the magnitude of this spring peak as a consequence of interannual differences in nutrient input from the watershed (Malone et al. 1988). Studies by Sellner (1993) and White and Roman (1992) indicated that the spring bloom was not extensively grazed by zooplankton. If it was deposited to deep waters, as seems likely, it would become available to support a host of processes including macrofaunal growth, microbial respiration, and associated oxygen consumption and sediment nutrient releases.

Deposition rates of total chlorophyll *a* were measured using fixed sediment traps (weekly or biweekly measurement periods) from 1985 to 1992 (Kemp and Boynton 1992; Boynton et al. 1994; Roden et al. 1995). Deposition from spring blooms (integrated from day 50 through day 150 in all years) ranged from 541 mg m^{-2} in 1989 to 1,190 mg m^{-2} in 1990. Estimates of spring-bloom deposition rates followed qualitative trends in algal biomass for some years but not others (Magnien et al. 1994).

Inspection of spring-bloom deposition rate and river flow data suggested that there might be a more consistent relationship between deposition and flow than to algal biomass during the spring bloom and deposition. A series of analyses was performed (Boynton at al. 1993) wherein the period of time during which flow was averaged was different (figure 11-5).

These analyses indicated that river flow that occurred just prior to the spring bloom had the most influence on the magnitude of subsequent deposition rates. Low river flow from December through February was always associated with small spring deposition rates as were freshets that occurred late in the spring after the time of normal spring-bloom development (for example, May 1989); the largest deposition events (1987, 1988, and 1990) were all associated with river flow patterns that featured a distinct above-normal pulse in flow from December through February. In this case, analyses suggested a close temporal coupling between flow and an ecosystem response. Spring-bloom deposition appears to be responsive to relatively recent river flow events, with integrated effects of flows from previous seasons not being evident.

A strong departure from the expected pattern was observed in 1991 wherein a relatively small deposition event was associated with very high and sustained river flows that began in fall and continued throughout the winter. This suggested that either the bloom did not develop or that deposition did not occur as usual. Water-column chlorophyll data suggested a strong bloom in 1991 (Magnien et al. 1994), which weakens the former hypothesis. It appears probable that the 1991 bloom deposited farther downstream than usual and thus was not measured at our fixed station. Maps of chlorophyll concentration in surficial sediments made immediately after bloom deposition in 1993 (another year with a strong freshet) indicated that most of the bloom deposited 20–30 km downstream of the sediment trap location. This analysis suggests that these systems are responsive to forcing events of relatively short duration and that the spatial location of the spring-bloom epicenter can be shifted seaward in years of high sustained river flow. These observations generally, but not always, conformed to the simple "bottom-up" control model presented earlier. The outlier in this data set was useful, as is often the case, leading us to consider

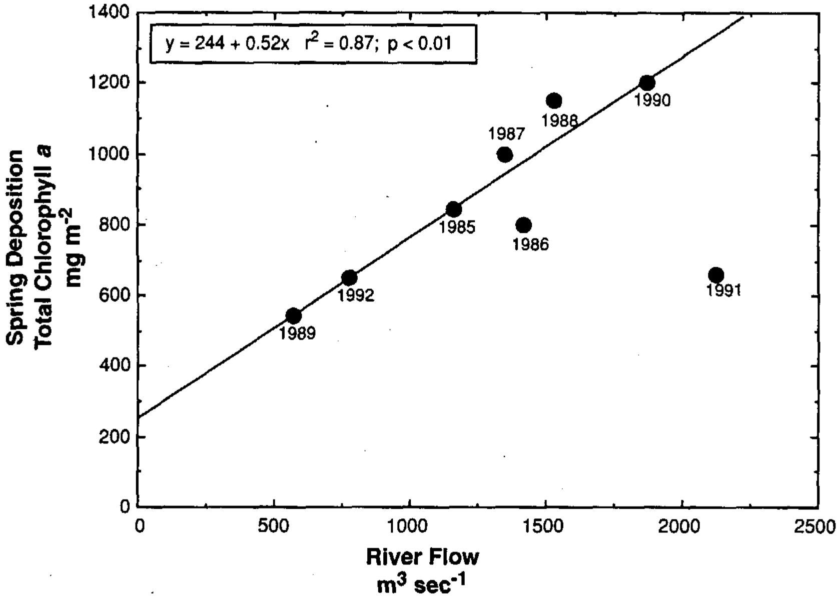

FIGURE 11-5 Results of linear regression model showing relationships between the magnitude of spring phytoplankton bloom deposition and freshwater inflows from the Susquehanna River. Spring-bloom deposition of total chlorophyll *a* was estimated from moored sediment trap arrays (duplicate weekly or biweekly measurements from collecting cups positioned beneath the pycnocline). Spring-bloom deposition was calculated as the total chlorophyll *a* mass collected from the initiation to termination of bloom deposition during each year. River flow was averaged for the months of December through March for each year preceding the spring bloom. All data are from the R-64 site. The 1991 data were not included in the regression model.

additional explanations that eventually resulted in a better understanding of these systems.

Seasonal Declines in Deep-water Oxygen

In many coastal areas, including Chesapeake Bay, there is strong interest in the scientific and management communities to better understand processes regulating dissolved-oxygen declines. In Chesapeake Bay low dissolved-oxygen conditions occur primarily in those areas where the water column is stratified and where there is substantial deposition of organic matter from the upper water column to deep waters and sediments. Establishing relationships between myriad environmental variables and oxygen conditions has not been a simple task in the bay. For example, Seliger and Boggs (1988) pointed out that low dissolved-oxygen conditions in the bay could be explained by river flow and water-column stratification and concluded that nutrient inputs (and the organic matter produced) may have little to do with anoxia. Kemp et al. (1992) argued that physical and biological processes are coupled, with freshwater flow providing buoyancy for stratification as well as nutrients for organic matter production. Boicourt (1992) has suggested the possibility of a phase shift in freshwater flow versus anoxia relationships.

Our data are not adequate to entirely resolve this problem but it is possible to test for relationships between deep-water oxygen characteristics and organic matter deposition rates for one region of the mainstem bay where seasonal oxygen problems are chronic. Several anecdotal observations indicated that such relationships might exist. First, hypoxic or anoxic conditions developed in deep waters for some period of time during each year since intensive monitoring began in 1984. Low oxygen concentrations in deep waters were associated with even the lowest flow conditions observed. Second, in 1989, the spring freshet (and associated nutrient load) did not enter the bay until mid-May. The spring phytoplankton bloom did not develop to any significant extent and deep-water oxygen depletion was delayed for about a month.

Finally, in 1992, the spring freshet was very small. Spring chlorophyll *a* concentration in the water column and chlorophyll *a* deposition rates were among the lowest on record and dissolved-oxygen concentrations declined slowly, not reaching mg l^{-1} until early July. These results suggest that deep-water oxygen conditions are regulated, at least in part, by the amount of organic matter deposited during spring.

Bottom-water oxygen concentrations were routinely measured (weekly or biweekly) at the R-64 station from 1985 to 1992 (figure 11-2). Water depth at this site is about 17 m and vertical water-column stratification is generally strong in this region of the bay. The daily rate of change of oxygen concentration ($d\,DO/dt$) was calculated using spring measurements from 1985 through 1992. The time period over which rates of change were calculated varied slightly among years but in most cases included the period from the beginning of March through the middle of May. The criterion used to determine the starting point was that the first observation should not be followed by any oxygen measurements of higher concentrations. Typically, during late winter and early spring, deep-water oxygen concentrations exhibit both small increases and decreases over time but are usually close to saturation. The final oxygen measurement used was the last measurement made before oxygen concentration declined below 1 mg l^{-1}. The rates of oxygen decline for the years 1985 through 1992 calculated from these data were linear, statistically significant ($p < 0.01$; $r^2 > 0.90$), and differed appreciably among years (by more than twofold).

The fact that dissolved-oxygen concentrations began declining during early spring suggested that these declines were caused by respiration of spring deposition events rather than later summer events. Accordingly, average spring deposition rates of total chlorophyll *a* were calculated for each year using deposition data collected between early February and the beginning of May. Chlorophyll *a* deposition rates were regressed against the rate of dissolved-oxygen decline derived from regressions of time versus dissolved-oxygen concentration (figure 11-6). These results suggest a strong influence of organic matter availability on the rate of oxygen decline. However, at least two alternative explanations exist. First, it can be hypothesized that different spring rates of oxygen decline are caused by interannual differences in temperature regimes. Oxygen decline would be more rapid in warm years than in cold years because of the influence of temperature on respiration rates (Sampou and Kemp 1994). This explanation seems unlikely to be the prime cause because interannual temperature differences have been small over the period of record. Additionally, warm and cool springs were not correlated with high and low rates of oxygen decline. The second hypothesis is that the cause is related to interannual differences in the strength of water-column stratification. In years when the water column is highly stratified, less mixing of oxygen from surface to deep-water occurs and rates of oxygen decline would be greater. Stratification certainly plays a major role in determining deep-water oxygen characteristics. However, the case for stratification being the dominant cause of interannual differences in oxygen decline rates is weak because years of high and low stratification do not correspond well to years of high and low rates of oxygen decline.

Sediment Ammonium Fluxes

The final example in this sequence concerns possible relationships between river flow and sediment nutrient recycling. It is hypothesized that variations in river flow and associated nutrient inputs regulate spring-bloom size and

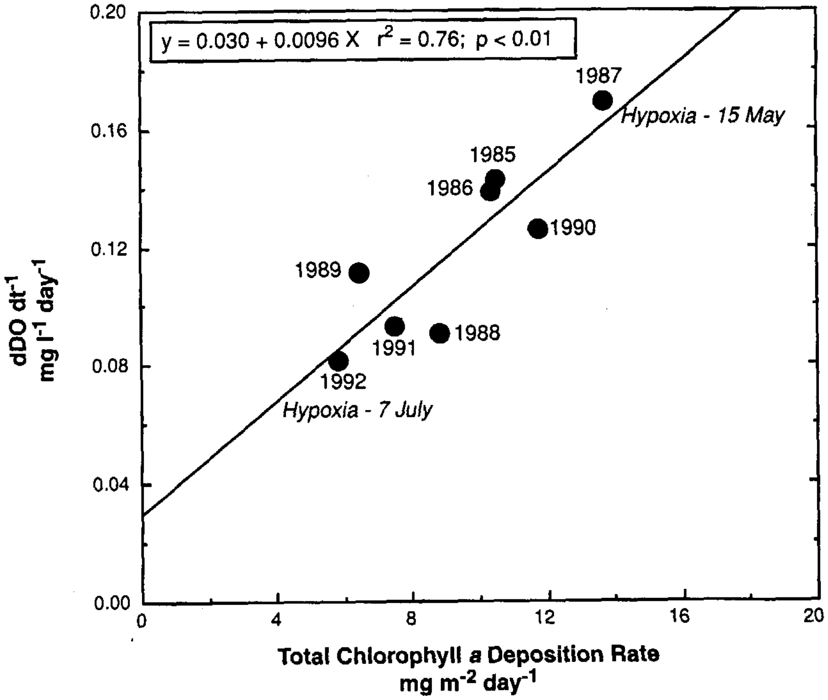

FIGURE 11-6 A scatter diagram showing the relationship between the rate of decline in dissolved-oxygen concentrations in deep water (*d*DO dt^{-1}) and average deposition rates of total chlorophyll *a* during the spring-bloom period. Data are from the 1985–1992 period and were collected at the R-64 site. The date on which hypoxia (DO concentration <1 mg l^{-1}) was first encountered during highest (1987) and lowest (1992) deposition years is also indicated.

organic matter deposition rates. Deposited organic matter, in turn, serves as a substrate for decomposers, which eventually regulates nutrient releases from sediments. We attempted again a series of regressions using different time averaging of flow and benthic nutrient recycling rates. Again, most combinations indicated a positive relationship; the strongest relationship between river flow and sediment ammonium flux was found using winter (December to February) flow rates, as in the deposition-versus-flow relationship, and summer (June to September) average benthic ammonium fluxes (figure 11-7). This implies a time delay between nutrient input and benthic nutrient recycling. In this estuary, springtime respiration rates remain relatively low at temperatures below 10°C for both benthic (Boynton et al. 1990) and water-column communities (Smith and Kemp 1995); rates increase exponentially with vernal warming beyond these thresholds. Deposition of organic matter to sediments derived from the spring bloom starts in late February and ends by

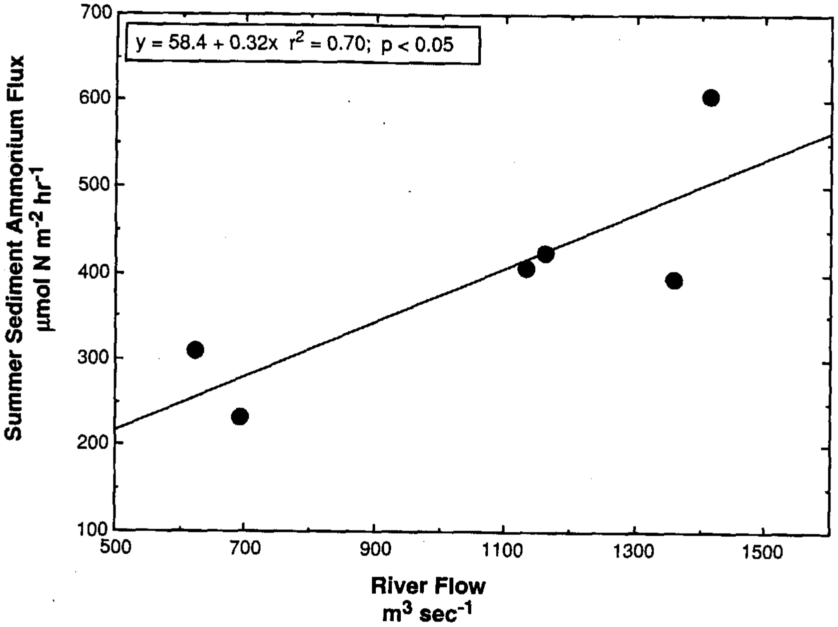

FIGURE 11-7 A scatter diagram indicating the relationship between summer sediment ammonium flux (June–September) and winter flow from the Susquehanna River. Sediment-water flux data are from the R-64 site and were collected during the 1988–1993 period.

mid-May. However, large sediment fluxes of ammonium are not evident until June when bottom waters are above 15°C and coupled rates of nitrification and denitrification begin to decline with oxygen depletion (Kemp et al. 1990). Relative to the other relationships presented here, the river flow-nutrient cycling relationship was the weakest. In part, this may be due to a more limited data set. It may also be because this process is the farthest removed from the influence of flow, at least as conceptualized here. In this view, other factors have more of a chance to come into play (for instance, infaunal community activities, sediment redox conditions, nitrification-denitrification, focusing of organic matter from shoal areas to deeper waters) modifying or fundamentally changing the nature of the flow-recycling relationship.

Intersite Comparative Analyses

In this section, we present additional examinations of estuarine features as they relate to variations in river flow and attendant nutrient-loading rates. The approach here is comparative wherein data from five sites with different nutrient-loading rates were used rather than a time series from a single site.

Here we use space (different sites) to examine the possible causes of temporal variability just as we used a time series of observations in the previous analyses (Pickett 1991). There are several distinct advantages to a comparative approach to synthesis. The range in both independent and dependent variables can be expanded if sites are chosen with this in mind, thereby increasing the possibility of observing patterns, if they exist. Additionally, the inclusion of multiple sites or different systems inherently increases the generality of conclusions; the possibility that observed relationships are only unique to a single site is diminished. However, difficulties present themselves with comparative analyses as well, the most prominent being the differences in the characteristic scales (such as volume, depth, residence time) among sites. Hence, there is a need to analyze data in a way that accounts for scaling differences so that ecological variables of interest are comparable among different systems. In fact, the use of nutrient-loading rate was adopted here because nutrient loads were known to be different among the systems we studied and could be scaled to the respective estuarine areas. The scaling of variables (such as nutrient-loading rate to estuarine area) is, in itself, an issue that could benefit from some serious consideration.

Nutrient Stocks

Perhaps the most direct relationship to consider is that between nutrient loads and water-column stocks (figure 11-8). Our analyses indicated a very strong relationship between annual average total nitrogen (TN) load and average annual TN mass in the water column; to a lesser extent the same was found for total phosphorus (TP). The weaker TP relationships may have resulted from the fact that a large percentage of the TP load is in the form of inorganic particulate phosphorus and hence not as prone to remain in the water column (Boynton et al. 1995). However, all results were poor when specific chemical species (such as nitrate) rather than totals were used. Apparently, specific nutrient species are transformed rapidly enough so that simple relationships to load are not apparent at that level of detail.

Another feature of these relationships is the attenuated response of concentrations to loads. For example, TN loads varied by a factor of about 10 while water-column mass varied by only a factor of 3. Similar attenuated responses were found for a variety of variables examined in a series of estuarine nutrient budgets (Boynton et al. 1995) and in a set of marine mesocosms exposed to a range of nutrient-enrichment rates (Nixon et al. 1986). This suggests that either internal sinks (such as sediment burial and denitrification) are quite active or that these nutrients are efficiently transported out of the system. In the case of Chesapeake Bay, both seem to be involved (Boynton et al. 1995). Finally, in some regression models the intercept values contain information of

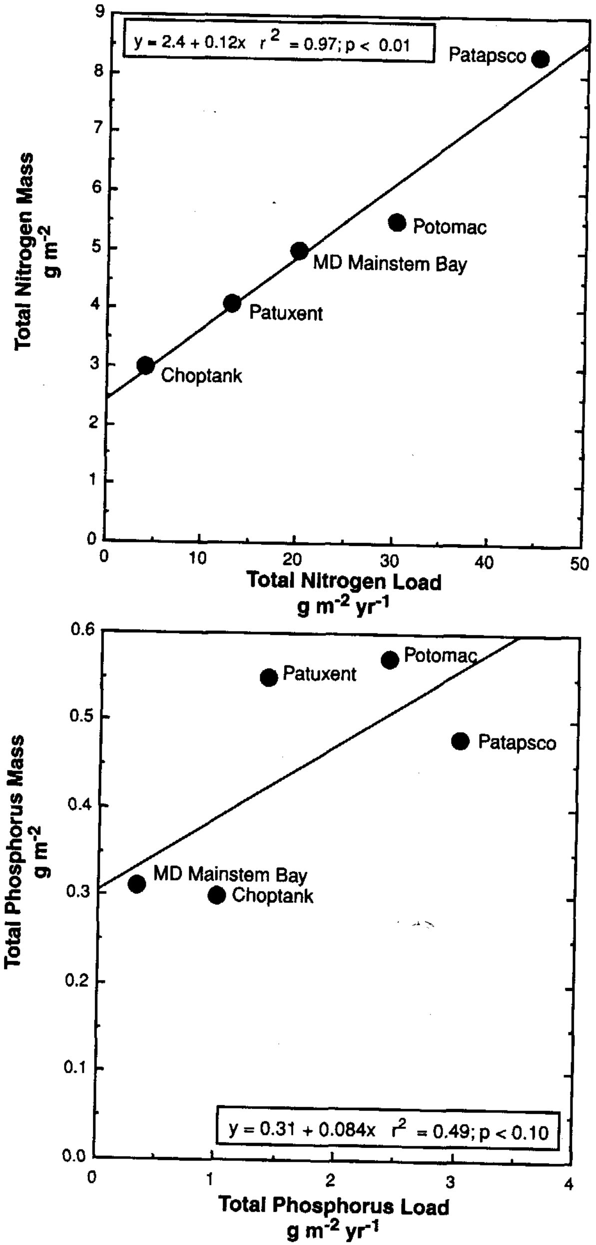

FIGURE 11-8 Scatter diagrams indicating relationships between annual average total nitrogen (TN) and total phosphorus (TP) mass in the water-column and average annual TN and TP loads to five locations in the Chesapeake Bay system. Loads and water-column masses were measured at the fall-line sites and estuarine stations, respectively, indicated in figure 11-2. Data were averaged for the 1985–1996 periods.

ecological interest. In this case, TN and TP values at zero loading rates are still substantial (~17 µM N and 1 µM P for a 10 m water column) and suggest potentially high productivity even under pristine conditions.

Phytoplankton Chlorophyll Stocks

In the mid-1970s limnologists developed a series of useful statistical models relating nutrient-loading rates and algal biomass for a large sampling of lakes (Vollenweider 1976). These relationships were used to estimate the degree to which nutrient-loading rates would need to be decreased to move a particular lake from one trophic state (as defined by chlorophyll *a* concentration) to another. Surprisingly, few comparable relationships have been developed for coastal and marine ecosystems (Nixon et al. 1986; Nixon 1988). We initially attempted a direct duplication of the Vollenweider (1976) model using average annual (or ice-free periods) surface-water chlorophyll *a* concentration ($\mu g\ l^{-1}$) as the dependent variable and annual average phosphorus loading rate (adjusted for the freshwater fill time and mean depth of the receiving water body) as the independent variable. This selection of variables did not produce either predictive or significant statistical results ($r^2 < 0.10$; $p > 0.10$) We then reasoned that, because algal blooms often develop in deep waters, particularly in spring in Chesapeake Bay, vertically integrated water-column chlorophyll *a* (mg m^{-2}) would be a better estimate of algal biomass; however, results were only marginally better. We then substituted nitrogen for phosphorus and results improved to the degree shown in figure 11-9. We have also obtained sufficient data to add results of the MERL eutrophication experiment (Nixon et al. 1986) and portions of Hillsborough Bay, Florida, data (Johannson 1991) to this analysis.

The results support the concept that, for some estuarine systems, phytoplankton biomass levels respond in positive linear relation to nutrient-loading rates. Further, there is some indication that different systems respond in a similar fashion when loading rates are scaled for local conditions of depth and flushing rates. This sort of analysis could be expanded to include other systems to explore the robustness of the relationship; a successful test would increase confidence in the conceptual model on which it is based. However, the conceptual model used here explicitly favors bottom-up control. It is almost certain that such a model would not work in instances where top-down controls become dominant, as in cases where intensive benthic grazing by introduced species (Alpine and Cloern 1992) or aquacultural activities are important (Meeuwig et al. 1998). Finally, the scaling of the nutrient load for estuarine flushing characteristics used in this example is primitive and would not be appropriate for estuarine systems with larger tides or limited freshwater inflows (Monbet 1992). More realistic formulations are needed. However, this is an example of where a synthesis activity clearly suggests some additional lines of inquiry; in this case, the scaling of important characteristics of ecosystems.

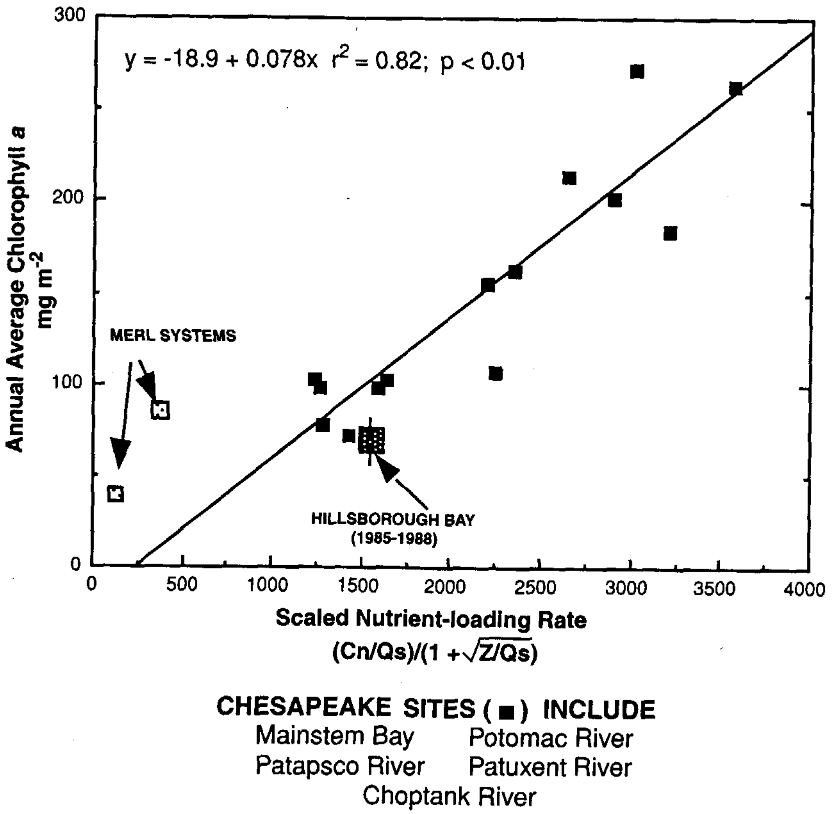

FIGURE 11-9 A scatter diagram relating annual average total chlorophyll *a* mass to nitrogen-loading rate. Data are from the five estuarine sites indicated in figure 11-3 and were collected during the 1985–1987 period. Nitrogen-loading rates were scaled following the method used by Vollenweider (1976) where: C_n = nitrogen-loading rate (mg N m^{-2} yr^{-1}); Q_s = hydraulic fill time (years); Z = mean depth (m). Hillsborough Bay, Florida, data are from Johannson (1991) and the MERL data are from Oviatt et al. (1986).

Sediment Nutrient Releases

In an earlier example, we related river flow to sediment nitrogen releases (figure 11-7) using time-series data. We considered the same processes again but used a comparative approach with data from several sites that encompassed a large range in total nitrogen-loading and sediment ammonium-recycling rates. As in the previous case, the conceptual model linking nutrient loading to the ecosystem from external sources and sediment nutrient recycling involved load-related algal biomass, which was deposited to sediments and eventually served as substrate supporting sediment nutrient releases. Results from one set of analyses are shown in figure 11-10.

The load-recycling relationship suggests several interesting insights. First, the slope of the regression indicates that for every unit reduction in TN load there would be about an equivalent reduction in sediment ammonium recycling. However, flux data are from summer when values are high; typical values from the remaining months are only 10–30% of these values. Overall, there still appears to be a strong linkage between load and flux. The intercept value of 120 μmol N m^{-2} hr^{-1} is sufficient to support relatively low rates of phytoplanktonic primary production (~0.3 g C m^{-2} day^{-1}, assuming Redfield C:N proportions). The intercept value would be lower if data from the Choptank River were excluded, as possibly they should be, because of problems with estimating the TN loads. At low nutrient-loading rates the amount of nitrogen recycled from sediments is small, but this could still be important in more oligotrophic environments.

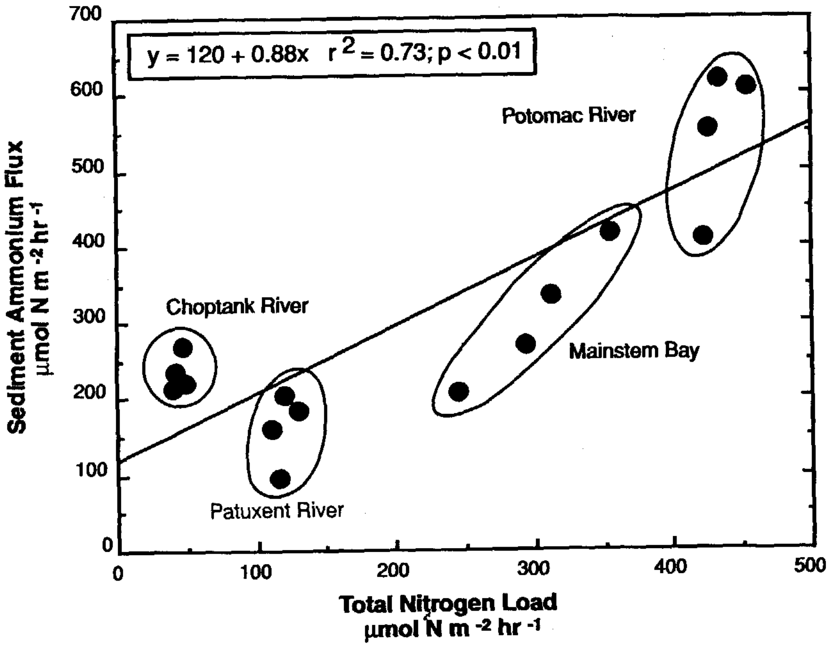

FIGURE 11-10 A scatter diagram relating summer (June through September) sediment ammonium flux to average annual total nitrogen load in four locations in Chesapeake Bay. Loads and sediment fluxes were measured at the fall line and estuarine sites, respectively, indicated in figure 11-2. Data were from the 1985–1988 period.

Choptank River fluxes are higher than expected for a given level of nutrient loading. This discrepancy may be more apparent than real. Results based on nutrient budget calculations indicate that the lower Choptank River receives substantial nutrient additions from the mainstem bay (Boynton et al. 1995). Groundwater discharges directly to tidal waters are also important. If this is the case, nutrient loading to the Choptank River is higher than shown in figure 11-10 and would have the effect of displacing Choptank River fluxes to the right, more in line with those of other systems. The fact that Choptank fluxes diverged so strongly prompted us, and others, to reexamine loads to this system, not an uncommon benefit of synthetic analyses such as these.

While only TN versus ammonium flux is shown in figure 11-10, other load-recycling relationships were examined. In general, sediment-water fluxes (for instance, sediment oxygen consumption, silica) were consistently better correlated with TN loading than with TP loading (Boynton et al. 1994). Even sediment recycling of PO_4^{3-} exhibited a stronger relationship with TN loads than with TP loads. In part, this may result from the fact that there is a considerably broader range in TN loads than TP loads (Boynton et al. 1995). It may not be possible to resolve TP influences on sediment nutrient recycling over this relatively narrow loading range. Alternatively, the poor correspondence with TP loads may indicate that most of the phytoplankton debris that reaches sediments, and eventually supports sediment-water nutrient recyling, was produced more in response to N than P availability in the water column. (D'Elia et al. 1986; Fisher et al. 1992).

There may be additional inferences to be drawn from this comparative analysis of TN loading versus sediment ammonium recycling (figure 11-10). Specifically, even in this limited (4 year) data set, there appear to be qualitative relationships of recycling to loading to each system; the pattern is most obvious for the mainstem bay site, but there is a hint apparent for most sites. The provocative observation here is that the slope of each cluster of points for 4 years tends to increase as ecosystem system size decreases. Thus, the fraction of TN loading that appears in summer benthic ammonium recycling is larger with small systems. This implies that smaller systems retain and recycle nutrients more efficiently, or that a larger fraction of primary production is deposited to and recycled in smaller (shallower) systems (Kemp and Boynton 1992; Boynton et al. 1995). Are there some rules for estuarine scaling to be gleaned from these types of observations? We intend to add more observations to this analysis in the future; it may well be that continued empiricism may provide answers to these questions.

Concluding Remarks

Examination of time-series and comparative data at sites in Chesapeake Bay have revealed surprisingly strong and linear relationships of primary production, benthic-pelagic coupling, and nutrient recycling to both freshwater flow and nutrient-loading rates. It appears that in well-sampled systems comparative analyses are not necessary to obtain sufficient range in variables for a pattern to emerge; in the case of Chesapeake Bay, clear signals were seen when river flow varied by about a factor of 2. However, comparative analyses increase the generality of results.

There are extensive data on water quality, physical forces, and ecological processes for Chesapeake Bay; we have only scratched the surface for inferences that could be drawn from these data. Data are available to explore the relations between freshwater flow (and nutrient loading) and distributions of seagrass, zooplankton, and benthic invertebrate communities. In addition, spatial sampling in many regions of the Chesapeake system is sufficient to develop volume-weighted estimates of processes, biomasses, and pool sizes; these variables would presumably be more representative of estuarine conditions than estimates based on samples from a single station.

The approach used here started with a conceptual model of how freshwater flow or nutrient loads influenced key processes and properties of an estuarine ecosystem. We have found these conceptualizations to be far more profitable than approaches that start with a "blind" search for correlations. However, most of our work has utilized simple linear regression techniques. More sophisticated techniques such as multiple linear and nonlinear regression, multivariate analyses, classification, and regression-tree and time-series approaches appear attractive because of the complexity of estuarine processes.

Based on our initial effort comparing ecological responses to variations in physical forces among Chesapeake Bay subsystems, we are optimistic about the utility of applying comparative analysis methods with time-series data for multiple ecosystems. We suggest that contrasting similar time-series analysis among different systems may help identify key scaling relationships needed to generate fundamental scientific understanding that is not site-specific.

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CHAPTER 12

Linking Biogeochemical Processes to Higher Trophic Levels

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Abstract

How well are we able to synthesize current understanding of how bottom-up controls affect higher trophic levels? We consider this question with respect to five general approaches: Long-term data collection, cross-ecosystem comparisons, computational models, nutrient and energy budgets, and experiments. The current status of each approach is assessed, published examples are identified, strengths and weaknesses are summarized, and recommendations are offered with specific attention to their utility for management. The foundation of estuarine synthesis is the formulation of models, both conceptual and mathematical. To the extent that models permeate this work, we emphasize the need to repeatedly question, revise, and test with data the fundamental assumptions of the models implicitly or explicitly underlying each approach.

Introduction

Synthesis: the assembly of parts into a unified whole. In estuarine ecology, this term has two consistent yet slightly different connotations depending at least partly on scale. First, information may be gathered in order to identify inconsistencies and to assess the relative importance of contributing parts. Most often, such synthesis applies to a single ecosystem such as a nutrient budget for an estuary. Second, comparable data may be assembled to identify patterns across time or space that may be of predictive value, or may suggest hypotheses about relationships appropriate for further study. A synopsis of seasonal chlorophyll levels in one estuary or a plot of production versus N loading in different systems develops a pattern that may predict behavior at different times or of other unstudied systems. These generalizations may be

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CHAPTER 18

Scientific Synthesis in Estuarine Management

Donald F. Boesch, Joanna Burger, Christopher F. D'Elia, Denise J. Reed, and Donald Scavia

Science and environmental management are, in a sense, relatively new acquaintances. To be sure, ancient Romans developed rudimentary understanding of the science and engineering necessary to manage the flow and disposal of sewage wastes. Eighteenth-century Londoners were acutely aware that something was rotten in the malodorous Thames—and that somebody had to do something about it. But, in general, the "solution to pollution was dilution," and serious scientific and technical solutions to environmental problems the world over only began to ensue well into the Industrial Age with the advent of secondary-treatment of sewage, and with a concomitant understanding that cause-and-effect relationships exist between human activities and environmental impacts. Even then, problems found solutions only on a case-by-case basis.

It was not until the period of heightened awareness of environmental problems, from 1969 to 1972, that the U.S. Congress passed a series of sweeping and ambitious environmental laws, such as the National Environmental Policy Act (NEPA) of 1970 and the Federal Water Pollution Control Act (FWPCA) of 1972, which committed the nation to approach environmental problems systematically and in doing so set a challenging example for the rest of the world. In some cases, these laws placed new demands on environmental sciences. For example, NEPA required lengthy Environmental Impact Statements that must summarize knowledge of the affected environment and assess the impact of the proposed activity and alternatives. But in other cases, such as the FWPCA, goals were set to eliminate discharges or control them to the limits of technology without regard to knowledge of actual impacts on the environment. Despite the many obvious successes in environmental management and pollution control in the United States, the fact that much of this seminal environmental legislation is based on uncertain science is now purported by some detractors to show that existing laws and regulations should be abolished.

For a great variety of reasons related to scientific uncertainty, the complexity of nature, the mismatch between the pace of science and the urgency of management decisions, differences in the cultures of practitioners, and political forces, there exists what has been referred to as a gap or "impedance mismatch" between science and environmental management (National Research Council 1995). This difficult interface is a source of frustration and complaint by scientists and managers alike. Nonetheless, modern management of coastal and estuarine environments is increasingly stressing the need for scientific knowledge. This is manifest in such programs as the National Estuary Program (NEP) and the Chesapeake Bay, Great Lakes, and Gulf of Mexico programs, and in the embracing of ecosystem management or integrated coastal management approaches (Healey and Hennessey 1994).

Toward the goal of improved development and use of synthetic scientific knowledge in estuarine management, in this chapter we first consider several cases studies of the use of science in estuarine or coastal management. We attempt to identify factors that have led to success and factors that have impeded it. We explore how synthetic scientific knowledge about estuaries can be developed and how it can be applied in site-specific management. Finally, we discuss the roles that federal and state agencies (both those that manage estuaries and those that support science) and the scientific community must play if we are to advance on this goal. Our premise is that environmental policies and regulations based on better science will be more effective and, accordingly, better regarded by all.

Accomplishments and Challenges

The examples we provide below are purely anecdotal: they have been chosen arbitrarily and consequently do not represent either a systematic or an exhaustive inventory of possible candidates for consideration. Nonetheless, we feel that they adequately represent a wide scope of situations in terms of both jurisdictional diversity and environmental complexity that valid general conclusions can be drawn from them.

Great Lakes

Probably the first national recognition of aquatic and coastal management problems occurred when the Cayahoga River at Lake Erie caught fire. While that event caught the public's eye, it was the extent of historically rich research results and the development and refinement of conceptual and mathematical models that refocused attention on the more pervasive overenrichment of the Great Lakes. This original focus on eutrophication gave way, after significant science-based management actions were enacted, to toxic contaminant, habitat, and fishery issues that were also identified primarily by the scientific community. Continued emphasis on mass-balance and modeling perspectives have provided effective interfaces between scientific discovery and management and policy action. Nongovernmental organizations (NGOs) have been very active and intergovernmental institutions have been developed to coordinate the jurisdictions involved. The International Joint Commission serves to coordinate science and policy between Canada and the United States. The EPA's Great Lakes Program coordinates state and federal activities within the United States. There is even a scientific society, the International Association of Great Lakes Research, which focuses heavily on the Laurentian Great Lakes and interacts closely with the IJC and other intergovernmental agencies (Francis and Regier 1995).

Today, the Great Lakes are hardly to be viewed as beyond environmental concern. However, there is clear recognition and acknowledgment that scientifically based actions initiated two decades ago have proved successful in confronting the problems identified at that time. For example, Lake Erie has benefited greatly from nutrient-control strategies that have been developed and is no longer in danger of severe eutrophication (Francis and Regier 1995).

Chesapeake Bay

As early as 1972 there were emerging concerns among the public and political leaders, boosted by some scientific observations and considerable speculation, about the effects of human activities on Chesapeake Bay. These concerns were heightened by the rapid population growth the region was witnessing. A Bi-State Conference on the Chesapeake Bay (Chesapeake Research Consortium 1977) identified the "continuing input of large quantities of chemicals which can be overenriching . . . baywide failure of oyster reproduction . . . and decline of rooted aquatic vegetation" as serious concerns and discussed population growth and point and nonpoint inputs of nutrients as possible causes. There was little hard scientific evidence of these linkages at the time, although later analysis of historical data did corroborate such an interpretation (Malone et al. 1993). Since then, scientists and managers have been engaged in a dynamic, occasionally contentious—but nonetheless productive—interaction that has resulted in major commitments of the multiple political jurisdictions in the Chesapeake Bay watershed for the treatment of point-source discharges, the control of nonpoint sources, and the use of increasingly sophisticated management tools, such as ecosystem models of the bay and its

Box 18.1

"Mature" Estuarine Management: Chesapeake Bay

Malone et al. (1993) provide a fascinating historical analysis of the role of science in the acceptance of nutrient overenrichment as a major cause of the degradation of Chesapeake Bay and in reaching commitments for nutrient controls. The Patuxent River, a relatively minor tributary of Chesapeake Bay that was the site of a naval skirmish and the British landing before the burning of the White House during the War of 1812, served as the site of early volleys in this metaphorical war to "save the bay." The watershed of this subestuary occupies only a small part of the 167,000 km^2 bay watershed but is located between the two largest metropolitan areas, Baltimore and Washington, and has been one of the fastest-growing areas in the United States.

Early interest in nutrient dynamics and primary productivity in estuaries at the Chesapeake Biological Laboratory (CBL), located near the mouth of the Patuxent estuary, led to studies that produced data on nutrients, water clarity, and dissolved-oxygen concentration in the estuary dating back to the mid-1930s. By the mid-1970s, concerns among CBL scientists about the declining water quality in the estuary and its apparent association with nutrient enrichment reached the attention of local government officials from the somewhat sparsely populated counties around the estuary, who were troubled by rapid population growth and increasing sewage discharges from the upstream counties.

Municipal waste treatment at that time focused mainly on secondary treatment and, following the successful experience in the nearby Potomac River estuary, on phosphorus removal. However, scientific evidence was growing that nitrogen was an important limiting nutrient in the brackish portions of the estuary. When the upstream counties, the state of Maryland, and the EPA all demurred on advanced nutrient removal, particularly nitrogen removal, a lawsuit was filed by the Tri-County Council of Southern Maryland, demanding substantial improvements in sewage treatment. CBL scientists testified on behalf of the downstream litigants, in opposition to the state and federal agencies sponsoring their research, and the suit was successful in forcing a consensus for advanced nutrient removal.

Also in the early 1970s widespread loss of submerged aquatic vegetation was observed by scientists at the Virginia Institute of Marine Science (in the lower bay) and the Horn Point Laboratory (in the upper bay). Segments of the public and scientific and management communities initially attributed these losses to increased agricultural herbicide use, major freshets, diseases, the disruptive activities of predators, or unassigned natural variability. However, greatly expanded research by VIMS and HPL scientists, stimulated by concerns over the vegetation losses, revealed that nutrient overenrichment throughout the bay ecosystem was an overarching cause of loss of submerged aquatic vegetation. Eutrophication had decreased light available to vascular plants growing in shallow waters by stimulating the biomass of phytoplankton and epiphytic growth.

Building on scientific discoveries concerning the Patuxent estuary and the demise of submerged aquatic vegetation, managers began to accept the notion that Chesapeake Bay as a whole is in trouble as a result of nutrient overenrichment. Very active NGOs such as the Citizens' Alliance for the Chesapeake and the Chesapeake Bay Foundation were successful in raising public awareness and concern. Senator Charles Mathias succeeded in the late 1970s in passing federal legislation to establish the Chesapeake Bay Program within the EPA. The Chesapeake Bay Program, under direction of its Executive Council including the governors of Pennsylvania, Maryland, and Virginia, the mayor of the District of Columbia, and the EPA administrator, is now a thoroughly institutionalized entity directing the restoration of the Chesapeake Bay ecosystem through coordinated nonpoint-source control, toxics reduction, habitat restoration, and living-resource management.

watershed (Boesch et al. in press). Considerable concern remains, but clear progress has been made (box 1).

San Francisco Bay

A small detachment of the U.S. Geological Survey is located on the shore of San Francisco Bay in Menlo Park, California. Here, an able group of federal scientists took a strong interest in their neighboring estuary and in the 1960s began to amass an impressive historical data set on the bay's water quality. They did much more, however. They analyzed the data with rigor on an ongoing basis and supplemented their analysis with a state-of-the-art program of process-oriented measurements. All of this information has been heavily used in the development of mathematical models of the bay.

Today San Francisco Bay is one of the better understood estuaries in the world. Fundamental information exists about seasonal and interannual variability in productivity, and there is keen awareness of the role of herbivorous

Box 18.2

A New Challenge: Florida Bay

In 1987, local fishing guides began to report that seagrass beds were dying throughout Florida Bay. Scientists confirmed these anecdotal reports indicating that by 1990 nearly 25,000 hectares of seagrass beds had been largely denuded. Although die-off events have been poorly monitored since 1990, current estimates place losses at more than 40,000 hectares. Florida agency and federal National Park Service scientists began relatively small-scale studies of the bay as a result of the public's concern, but resources directed to these studies were inadequate for gathering sufficient data to establish cause-effect relationships. No policy shifts were seriously contemplated at the time.

By 1990, more than 1,500 km^2 of massive algal blooms and high levels of turbidity were observed, and public attention to Florida Bay increased substantially. The first blue-green algae blooms appeared in the fall of 1991 following an extensive seagrass die-off event. These fundamental changes in the Florida Bay ecosystem have affected the abundance of important fishery species. Since seagrass beds are critical habitat for juvenile pink shrimp and Florida Bay is a major nursery ground for that fishery, the seagrass die-off was believed to be the primary contributor to a precipitous 50% decline in pink shrimp harvest and annual losses of more than $10 million.

Scientists began to speak out—somewhat reluctantly at first—on potential causes of the seagrass die-off and algal bloom problems. Hot and public debates, fueled by the media's penchant for the dramatic, have occurred among prominent scientists whose explanations of causality varied from freshwater diversion resulting in hypersalinity-induced mortality of seagrass to algal blooms induced by increased nutrient input from South Florida agriculture and leaching septic fields in the Florida Keys. Concern about the effect of water diversions on the Everglades led many in the public to conclude that decreasing freshwater input was the major culprit, but, in truth, studies have not been undertaken to provide full confirmation for any of the alternative explanations proposed within the scientific community. Bewildered public advocates have shaken their heads in dismay at the fractious scientists' inability to provide supportable answers. These scientists, in turn, were unable to find adequate support to fund necessary process-oriented research.

In 1992, the NOAA Florida Keys National Marine Sanctuary was created by Congress. The sanctuary itself was controversial in the public's mind, because it portended that use and access to natural resources would be regulated. Ironically, this controversy increased awareness and elicited further public outcries of concern for the Everglades, Florida Bay, and the Florida Keys. This, in turn, led to widespread and open speculation that these systems were somehow linked and must be managed collectively.

Serious attention is now focused on gathering more information on which scientific conclusions and adequate public policy could be based. State and federal agencies, including the National Park Service, the USGS, NOAA, EPA, the Florida Department of Environmental Protection, and the South Florida Water Management District, are committing several million dollars per year to research, monitoring, and modeling in Florida Bay. Interagency coordination of this effort is, of course, a formidable challenge, but the opportunity exists for rigorous yet management-relevant science.

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bivalves in checking the growth of phytoplantonic blooms in the heavily nutrient-enriched South Bay. Most concern rests on the effects of freshwater diversion from the bay to supply agriculture in the San Joaquin Valley and urban populations in Southern California. These strong competing interests have, until recently, overwhelmed considerations for supplying sufficient flow into the estuary to support fisheries production (Boesch 1996).

Florida Bay

In contrast to Chesapeake and San Francisco Bays, scientific research in Florida Bay has been relatively modest. It is remote, few people live on its shores, and attention has historically focused on the noteworthy national resources of the Everglades to the north and the Florida Keys to the south. With a limited base of scientific understanding, managers have been challenged to respond to the major changes that have taken place in Florida Bay only within the last decade (box 2). These changes include the die-off of vast areas of seagrasses, the unprecedented eruption of algal blooms in the bay's previously clear waters, and the impacts of these phenomena on fishery resources (Boesch 1996; Fourqurean and Robblee 1999). Diversions of fresh water from the Everglades to the Atlantic Coast and resulting increases in salinity in the shallow evaporating pan that is Florida Bay are thought to underlie many of the changes in the bay, but controversies have raged among scientists over exactly what the causes are. Although some of the changes now witnessed were predicted by scientists more than 20 years ago, lack of background knowledge—for example, the phytoplankton and its relationship to nutrient supply were largely unstudied—has slowed management response.

Because of newly heightened concerns, much greater attention by federal and state agencies is now being focused on Florida Bay and South Florida in general. Support for scientific research and monitoring has been greatly increased, presenting an opportunity to approach key management questions and scientific hypotheses strategically, relatively free from the limitations imposed by the entrenched positions of institutions and "experts."

Factors Leading to Success

Since no problem would ever reach the public and media consciousness without an active and vocal citizenry, it is clear that the role of NGOs is crucial at the outset to bring attention to problems and pressure to bear on public officials. However, NGOs alone do not normally possess the scientific expertise necessary to identify causes and effects and thereby develop the best prescriptions to solve the problems. In addition, there are other factors, some of which are often overlooked, that we believe lead to success:

1. *Key individuals* have, at the right time, helped to bring together those with a stake in a problem and those with the means to correct it. Not uncommonly, these individuals have been civic scientists (Lee 1993) with a personal interest in solving the problem. In several cases, active research institutions have existed for some time on the banks of the coastal water body in question, and the key individuals have been on that institution's staff. Sometimes these individuals have been public officials who took interest in the relevant scientific problems and who maintained a visible and active presence in public life for a decade or more.

2. A *lead agency* has been identified either by assignment or by acclamation or has for one reason or another emerged from among those having regulatory or managerial responsibility.

3. An *institutional structure* was developed for management that included a range of stakeholders—scientists, the public, agencies, conservationists, and technical experts. Although litigation or political controversy may have been initially responsible for the development of this structure, good will has furnished its long-term underpinning. This institutional structure must have access to and use the best scientific and technical advice available.

4. *Long-term scientific data* were available to document clearly that a problem had developed. Typically this was not the result of design: it just happened that a research laboratory located on or near the site of interest accumulated monitoring and process-oriented data that have proved useful in documenting change and the underlying causes.

5. A *visual environmental event* or *widespread public perception of a problem* galvanized official action. Some key event has occurred, such as the seagrass die-off in Florida Bay, that focused attention on the problem.

6. *An ecosystem-level view* of the system was taken, including the watershed as well as the coastal water body. This has led to better conceptual modeling of potential problems, promoted the view that many people have a stake in both the causation and impact of environmental perturbations, and involved multidisciplinary groups of scientists, social scientists, and engineers.

Science has played its most effective role when there was a large background of scientific information and sustained scientific investigations that led to an understanding of the system; a significant intellectual and logistical capacity in the regional scientific community that could be used to help refine management objectives; and a tradition of management-relevant research within that community (see also Boesch 1996). In Chesapeake Bay, both Maryland and Virginia had supported research institutions working on the environment and living resources of the bay for many years before the broad degradation of the ecosystem was identified and accepted. Even then, development of a scientific consensus, acceptance that the bay was generally overenriched, and commitment to restoration was anything but a smooth process (Malone, et al. 1993).

Long-term studies such as the Chesapeake Bay monitoring program, investigations conducted by the USGS and the California Department of Fish and Game in San Francisco Bay, and other regional environmental monitoring programs have advanced our understanding of estuarine systems. In the Great Lakes, U.S. and Canadian scientists from both the government and the university community built an impressive data and knowledge base from the 1960s through the 1980s. This information helped not only to identify and clarify extant and emerging issues, but also to evaluate potential results of proposed management actions. More recently, sustained investigations through such initiatives as the Land Margin Ecosystem Research Program and the NOAA Coastal Ocean Program are contributing to our fundamental understanding of processes that must underpin effective management decisions.

Factors Interfering with Success

The experiences related to the role science played in the previously mentioned examples of success, together with observations of less successful environmental management efforts, allow us to identify some common factors limiting the development and use of scientific information and understanding to manage coastal ecosystems more effectively.

Many coastal regions of the United States, including several that are experiencing severe environmental degradation, do not have such an extensive body of background information or as large and diverse a regional capacity for science. Florida Bay, for example, has not been nearly as well studied as the Chesapeake, San Francisco Bay, or the Great Lakes. Moreover, there are relatively few investigators working in that system. How, then, can scientific understanding be quickly and efficiently developed to guide the protection and management of this important coastal ecosystem? What understanding can we extend from other areas to contribute to this process?

Of course, a limiting factor in the advance of science is financial support. Too often, programs may be well supported for a short period of time but not sustained. Too often, programs have focused on bits and pieces of the problem but have not been comprehensive. Very important, funding for scientific synthesis, including analysis rather than just a literature review, has been scant. This is particularly so for syntheses that include diverse estuarine systems and offer the prospect for development of more generic understanding of estuarine processes that could be broadly useful for ecosystem management.

When monitoring data have been collected, it has often been without a clear sense of purpose or potential applications (National Research Council 1990). Accordingly, in many cases data quality has been inadequate for the interpretive need. In other situations, "monitoring" has been confused with "research." Process-oriented studies have not been conducted that are essential to understand system behavior and response to stressors.

A confounding problem has been the plethora of federal, state, and local agencies, as well as research institutions and departments, involved in estuarine management. Among the federal agencies, EPA, NOAA, DOI, the Army Corps of Engineers, and USDA are involved in most regions. At the state level, environmental protection and natural resource management are usually the responsibility of separate agencies. With the same agencies, inland environments and resources may be the responsibility of different bureaus than estuarine and coastal environments and resources. This poses a significant challenge, if not an outright obstacle, to ecosystem management of estuaries and their watersheds. Environmental management programs have been more or less successful depending on the degree to which we have been able to break down or tunnel through these jurisdictional walls.

Walls also exist within the scientific community, between disciplines and between media (estuarine, terrestrial, freshwater, atmospheric). The scientific community has played a major role in helping managers understand the connections among the estuary, the watershed, and the atmosphere. But it has been less successful in making the needed connections to aid understanding of the quantitative couplings among the environmental media. This is at least partially because estuarine scientists are generally located in different institutions or departments than experts on watershed and atmospheric processes. Although we are beginning to have some success in the needed intermedia integration, there is still a long way to go.

Developing Synthetic Scientific Knowledge

Synthesis can be motivated by one of two general needs. The first is a desire to explore and reveal fundamental properties of systems, properties that represent common attributes or underlying truths about how systems function in general. The second motivation for synthesis is to attempt to "scale up" local information to propose testable mechanisms for predicting system behavior in general. While both motivations are valid and, in some cases, coincident, the latter approach leads more directly to filling the niche between scientific discovery and management application.

Other sections of this book deal with specific aspects of this scaling up for particular components of system structure and function. Here we discuss scientific synthesis, including scaling of ecosystem structure and function in management applications, which occur primarily at local or regional scales.

Synthesis in this context almost always has to be based on a comparative analysis. That is a cross-sectional analysis that generally relates system drivers to system structure and function (for example, algal biomass as a function of nutrient loads, fishery yield as a function of algal production, denitrification as a function of nitrogen load) or in the development and application of simulation models across systems. As is discussed in other parts of this book, the products of such analysis can include conceptual models that illustrate key interactions, bivariate or mulitvariate correlations, nondimensional analysis, and a wide range of simulation models.

While these types of analyses have been developing slowly over the past decade for estuaries, generic simulation models and cross-sectional analyses for lake systems have a long and well-documented history (e.g., Vollenwieder 1976). While intellectual debate continues about the relative benefits of understanding detailed mechanisms behind the more empirical comparisons, the history and development of cross-system comparative analysis has clearly provided freshwater policymakers and managers with credible tools to deal with important problems.

What did limnologists have then that estuarine scientists perhaps do not have now? We suggest the answer may be "data and information from a plethora of systems." From the mid-1960s to the mid-1980s, there was a wealth of lake studies (from monitoring to detailed process studies and model development) on which to build *and test* models and cross-sectional tools. Healthy competition among modelers and "synthesizers" and the availability of a rich data set from a wide range of lake types led rather quickly to a new level of systemic understanding and growing predictability of how lakes function and how they are likely to respond to particular management action.

It is unlikely that the limnological experience can be repeated for estuaries. Not only is there generally less funding available for these efforts, but estuarine research and analysis are typically more expensive. However, there is substantial support for estuarine study and significant progress is being made toward the development of synthetic conceptual and numerical models of estuarine processes. With the proper overall framework and a community consensus on need and direction, there could be greater financial support *and* we could make more progress toward a synthetic understanding of estuarine systems using the support that exists.

When looking only at federal programs devoted to research and science-based management of estuarine systems, one finds an impressive array of efforts. For example, EPA supports efforts on twenty-eight bays and estuaries through its NEP and Chesapeake Bay Program; NSF has supported research efforts in six estuarine and coastal sites in its LMER and LTER programs. NOAA supports the establishment and operation of twenty-five estuarine research sites through its National Estuarine Research Reserve System (NERRS) and seven regional coastal and estuarine ecosystem studies through its Coastal Ocean Program (COP). Most of these programs are designed to build on important individual efforts by principal investigators supported by NSF and the National Sea Grant College Program and attempt to move focused efforts into larger ecosystem perspectives. While these efforts can never cover all estuaries, they do represent a significant range in type and diversity in drivers to form the basis of a comprehensive comparative assessment and synthesis.

Greater emphasis should be placed on coordination and cross-comparison of studies from these existing programs. This could be achieved essentially within the existing research framework through specific funding for comparative studies and synthesis efforts.

Applying Synthetic Scientific Knowledge to Site-specific Management

We have identified several instances in which scientific understanding of ecosystem function has been the basis of management plan development and management implementation has benefited from the application of this

knowledge. The examples we describe are from large estuarine systems where, even before the identification of an ecosystem problem, nationally recognized scientists had been successful in attracting funds to conduct basic research. This research, conducted at the scale of the system within which the ecosystem problem was later identified, was then directly transferable to management agencies and personnel with responsibilities for those specific estuaries. We recognize that funding of basic research within individual estuaries may not be the most effective way of providing scientific understanding toward ecosystem management and the solution of estuarine problems, and that synthesis efforts may provide a more useful framework for using science to manage estuaries more effectively.

Our challenge, to take our understanding of estuarine science to a higher level by synthesizing across systems and recognizing fundamental aspects of estuarine function that operate in systems of all scales, brings with it an additional challenge: *to provide and translate our new synthetic understanding of estuaries to the local level where it can be utilized in the formulation of management strategies.* The motivation behind synthesis efforts in ecosystem science is to expand understanding gained from a few intensively studied systems to provide information that can be applied more widely. In a management context such a mission calls for applying lessons learned from our research in selected systems, where research programs have been focused, to management issues in smaller and/or less well understood coastal systems—from Chesapeake Bay and San Francisco Bay to Matagorda Bay and Tillamook Bay. As we scale up our scientific understanding, we must be sure that the principles we recognize can be applied to systems of all scales.

One approach to this challenge is the development and refinement of general models that can be utilized in estuaries of various sizes with the input of certain information about the local system. These models may link processes in the watersheds draining to estuaries and in the coastal ocean to the processes in the estuaries themselves (NRC 1994b). For example, models have already been developed that can successfully simulate the development of hypoxic and anoxic conditions that result from excessive nutrient loading in Chesapeake Bay (Boesch et al. in press). These models have been translated to simulate the operation of the same ecosystem functions in other systems with some degree of success. So far their application has been to systems where hydrodynamic models have either been available or readily formulated and information about watershed inputs is also available. In addition, operation of these complex models requires a high level of technical expertise. The use of such modeling approaches to the formulation of management strategies in less well-understood systems would require (1) a baseline level of information about the estuary; (2) the formulation of various models to address the variety of aspects of ecosystem function that can be the foundation of management problems; and (3) the development of user-friendly interfaces.

An important component of the application of such models to estuarine planning and management efforts is the transfer of information about the model from the modelers with technical knowledge of estuarine processes to the managers who will use the model to assist with management decisions. Using ecosystem models requires model-specific knowledge of what the model includes, what it can be used for, and its limitations. This level of understanding is not typical amongst the management community but efforts can be made to develop self-documenting models designed for use by the management community. Such approaches require dedicated input by technical experts, usually within the research community, to develop the model and oversee the generation of management scenarios and the use of model output by estuarine managers. The provision of a model to aid with management does not negate the need for ongoing dialogue between estuarine managers and the scientific community.

Successful development of models keyed to assisting with management decisions might require identification of baseline physical, hydrographic, biological, and chemical data that would be required for model application. The scientific community could use existing models to identify minimum levels of resolution for data collection for estuaries of different scales. Local resource and management agencies would collect and manage these data for their estuaries. It might also require development of models that address a variety of estuarine management problems, for instance, secondary productivity and loss of emergent or submerged vegetation habitat.

The successful application of such problem-solving models requires clear recognition of the scientific basis of the management problem. A model that assists estuarine planners and managers in making decisions about alternate ways of managing nitrogen input to an estuary should not be applied until nutrient loading has been identified as a management problem. The level of scientific understanding necessary to identify the problems should still be concentrated within the research community.

In addition to these efforts, we need to develop and synthesize experience gained from the application of science to management issues in more than twenty-nine estuaries included in the NEP and Chesapeake Bay Program. The mission of these programs is to document and prioritize environmental problems within the estuary and to develop management strategies for these problems based on technical information and understanding. This process usually involves scientists with local knowledge of the estuary as well as resource agencies and managers with responsibility for managing the estuary. The NEP provides an interface between these groups and in many cases constructive dialogue regarding approaches to systems management and problem solution has resulted. This "bottom-up" approach to estuary management is based on individual estuaries; however, some synthesis and recognition of successful approaches across the NEPs may provide valuable insights into the relationship between our scientific understanding of estuaries and the development of management strategies.

Progress in this area can be achieved by conducting a synthesis of management experience across the NEPs with a focus on the role of science in the formulation of management policy, and the success of those policies once implemented. The synthesis would be developed through a review of NEP Comprehensive Conservation and Management Plan (CCMP) documents, to identify direct use of scientific understanding in the management plans, and a workshop with scientists and managers from each NEP who have been intimately involved in the development and implementation of the CCMPs.

Role of Federal and State Agencies

State and federal agencies have played an important role in the development and synthesis of estuarine and coastal science and will continue to do so. These agencies have a unique role in continuing to fund, direct, encourage, and in some cases participate in basic science, methods and tool development, and synthesis across problems and estuarine systems. Coordination between state and federal agencies, as well as local and regional agencies, is essential for enhancing the interface between science and management. This coordination and communication should include a number of stakeholders, including scientists and agency personnel at all levels.

Some federal programs for basic research have both significantly advanced science and provided insights useful for managing coastal ecosystems. The LMER and LTER programs, funded by the NSF, the federal basic research agency, are two that have bridged the gap. Both programs have supported sustained research on key environmental processes. Expanding such efforts to cover a wider variety of estuarine ecosystems would greatly enhance the capability to bring synthetic knowledge to bear in estuarine management. One program that has sought to support innovative science relevant to key management concerns is NOAA's Coastal Ocean Program (COP; National Research Council 1994a). It involves managers in the definition of critical scientific questions and engages both federal and academic scientists working together to answer those questions. Should it succeed, the COP may be a model for future development of federal programs.

Several federal programs are aimed at monitoring and assessing estuaries and other coastal ecosystems on a nationwide basis. These include the NEP, National Status and Trends Program (NSTP), and the Environmental Monitoring and Assessment Program (EMAP). Although broad-scale monitoring programs such as NSTP and EMAP have provided a basis for interregional comparisons, information from these national assessments is not extensively used in local or regional ecosystem management either because the sampling intensity (spatially and temporally) is too coarse or measurements specifically relevant to the ecosystem in question are not made.

On the other hand, monitoring or scientific assessment within localized management programs (such as the NEP sites) may lack national standardization, thus impeding interestuary comparisons and synthesis. The heavy local focus of these programs, while improving relevance of information to the estuary in question and involving local experts and stakeholders, misses opportunities for extending the knowledge of other estuarine ecosystems to the case at hand. In that sense, the relative detachment of the scientific assessments at the twenty-eight ongoing or completed NEP sites has not allowed us to advance knowledge in a way conducive to its ready exchange among these ecosystems or its extension to less well-studied estuaries.

A major institutional challenge to developing efficient working relationships in estuarine science and management is the large number of responsible federal, state, and local agencies with which scientists may need to interact. Jurisdictions and responsibilities are divided or overlap in complex ways. For example, NOAA has responsibilities for endangered sea turtles while they are in the water, while the U.S. Department of the Interior has responsibility for sea turtles when they emerge on beaches to lay eggs. States are responsible for fishery stocks that reside fully in estuaries, but responsibility for species that migrate between estuaries and the continental shelf or among estuaries may be shared with the National Marine Fisheries Service or regional fishery management councils. Within many states responsibilities for environmental protection, fisheries management, and coastal-zone management are in different agencies. Even when, as in New Jersey, they are contained in a single agency, responsibilities relevant to estuarine management may be dispersed among several bureaus (table 18-1).

Recommendations for federal and state agencies to foster synthesis and the application of synthetic knowledge in estuarine management include:

- Designating within each local or regional estuarine management program a lead agency with responsibility for taking the broad view of ecosystem management and coordinating the contributions and responsibilities of the other agencies

- Encouraging and implementing within each estuarine management program communication among stakeholders at all levels of government as well as scientists and the general public

- Supporting an ongoing scientific advisory process: charging the scientific

TABLE 18-1

An example of the division of responsibilities with respect to coastal environmental management within one state agency, the New Jersey Department of Environmental Protection.

Division/Office	Responsibility
Environmental Regulation	
Hazardous Waste	Hazardous waste permits
Land Use Regulation	Watershed management, tidelands management
Land & Water Planning	Watershed management, coastal-zone management, coastal nonpoint pollution control
Pollution Prevention	Source reduction of hazardous materials
Air Quality Regulation	Review of new proposals
Wastewater Facilities Regulation	Regulation of discharge of toxics, discharge permits
Natural & Historic Resources	
Fish, Game and Wildlife	Protection of endangered and nongame species, evaluation of effects of development, monitoring of wildlife diseases, regulation of marine fish harvest
Engineering and Construction	Planning of shoreline protection projects, maintenance of waterways, cooperation with Corps of Engineers, harbor cleanup
Green Acres	Assistance to counties for land acquisition
Enforcement	
Field Operations	Air and environmental quality enforcement
Enforcement Coordinator	Coastal and land-use enforcement, cooperative coastal monitoring, local management
Policy & Planning	
Environmental Safety, Health & Analytical Programs	Enforcement of state and federal pesticide laws, prevention of hazardous-material spills
Science & Research	Risk assessment/risk reduction, water-quality monitoring, monitoring of shellfish waters
Solid Waste Management	Issuance of landfill permits, waste-flow surveillance
Air Quality Management	Monitoring of air pollution from mobile sources, global warming issues
Energy	Implementation of master plan, planning of energy development and distribution compatible with protecting environment and public health

advisors with the task of synthesizing knowledge about the particular estuarine ecosystem and placing this in the context of estuarine ecosystems elsewhere

- Providing sustained support at the national level for fundamental research on estuarine processes within a framework of comparisons among ecosystems through such programs as the LMER the COP.
- Emphasizing comparison and synthesis within national networks of locally focused programs such as the NEP and the Sea Grant and Coastal Zone Management programs. The goal should be to develop generalized models that can be used by estuarine managers who have to deal with highly variable scientific knowledge for the estuaries they are managing.

Role of the Scientific Community

In striving to improve the application of scientific information and understanding scientists are generally quick to focus on the needed improvements in research funding and in the management sector and less ready to contemplate changes needed within the scientific community itself. But it is clear to us that scientists must also improve their efforts to synthesize results and communicate scientific understanding to managers. Some suggestions for improvement are listed below:

- The scientific community should improve the definition and articulation of important issues. Too often, efforts to bring the scientific community together to assess a problem and needed corrections end up with producing a long laundry list rather than a focused appraisal. Scientists should develop better processes to provide focus on the most serious issues, the most critical unknowns, and the most important priorities. The scientific community should also work to improve the process by which such information is clearly articulated to managers, policymakers, and the public.
- Scientific institutions should change the reward system for scientists, to providing more recognition of the value of contributions to synthesis and application as legitimate forms of scholarship (Boyer 1991). Such a change will need to be cultural as well as administrative, because this reward system depends heavily on peer recognition. Professional societies should take initiatives toward effecting these cultural changes.
- More leadership at the interface between science and management is required from scientific institutions and their administrators. This should include facilitating the efficient and effective involvement of institutional faculty and researchers, so that it does not become a time sink that slows the progress of science. It should also include follow-up to ensure that the contributions scientists make to agency deliberations are considered and used.

- Watershed/estuary-scale ecosystem science should be advanced by more concerted efforts to promote dialog and collaboration within the estuarine, freshwater, and terrestrial environmental science communities.
- Institutional arrangements wherein scientific information and advice are transferred should be reexamined and, where needed, redesigned to promote more effective use of synthetic knowledge about estuaries. This includes scientific advisory committees for regional management programs (such as the NEP's) and extension activities (for example, through the Sea Grant College Program).
- Scientists should promote and lead the application of advanced communications technology for the use of synthetic scientific knowledge in estuarine management. The academic and research community are very effectively using the Internet for individual and group communications (via e-mail) and for information dissemination (via the World Wide Web). The scientific community can be a catalyst for wider spread and more effective use of this technology in communicating scientific information and understanding to managers. Particularly important would be advancing the access to distributed databases through metadata servers. This could result in dramatic improvements in effectiveness. In addition, universities are leading the applications of multimedia communication, including interactive video, for teaching, teleconferencing, and the like. This offers considerable improvement in effectiveness and efficiency (saving time in traveling to meetings) at the science-management interface. In addition, the growing ease of telemetry in transmitting data could make similar contributions to the advancement of environmental research and monitoring.

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