Chesapeake Bay Eutrophication: Scientific Understanding, Ecosystem Restoration, and Challenges for Agriculture

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ABSTRACT

Chesapeake Bay has been the subject of intensive research on cultural eutrophication and extensive efforts to reduce nutrient inputs. In 1987 a commitment was made to reduce controllable sources of nitrogen (N) and phosphorous (P) by 40% by the year 2000, although the causes and effects of eutrophication were incompletely known. Subsequent research, modeling, and monitoring have shown that: (i) the estuarine ecosystem had been substantially altered by increased loadings of N and P of approximately 7- and 18-fold, respectively; (ii) hypoxia substantially increased since the 1950s; (iii) eutrophication was the major cause of reductions in submerged vegetation; and (iv) reducing nutrient sources by 40% would improve water quality, but less than originally thought. Strong public support and political commitment have allowed the Chesapeake Bay Program to reduce nutrient inputs, particularly from point sources, by 58% for P and 28% for N. However, reductions of nonpoint sources of P and N were projected by models to reach only 19% and 15%, respectively, of controllable loadings. The lack of reductions in nutrient concentrations in some streams and tidal waters and field research suggest that soil conservation–based management strategies are less effective than assumed. In 1997, isolated outbreaks of the toxic dinoflagellate *Pfiesteria piscicida* brought attention to the land application of poultry manure as a contributing factor to elevated soil P and ground water N concentrations. In addition to developing more effective agricultural practices, emerging issues include linking eutrophication and living resources, reducing atmospheric sources of N, enhancing nutrient sinks, controlling sprawling suburban development, and predicting and preventing harmful algal blooms.

EUTROPHICATION—an increase of the rate of supply of organic matter (Nixon, 1995)—has probably been more extensively studied in the Chesapeake Bay than in any other coastal ecosystem. Scientists have uncovered the sources of nutrients, how they stimulate biological productivity in the Bay, and how eutrophication results in oxygen depletion (hypoxia), increased turbidity, loss of submersed vegetation, and alteration of food webs. Furthermore, the multistate effort to restore the Chesapeake Bay ecosystem by reducing the inputs of nutrients that stimulate organic over-enrichment is one of the world's most ambitious attempts at large-scale ecosystem restoration.

The Chesapeake Bay is the largest estuary in the United States, with a length of more than 300 km and a total area of tidal waters of 11 000 km^2 (Fig. 1). Its 167 000 km^2 watershed extends over six states and the District of Columbia and accommodates a human population of more than 15 million. The Bay is a highly productive estuary, once called "the immense protein factory" by H.L. Mencken. Its economic and social importance to the region and its proximity to the nation's capital have long commanded special attention.

For most of the 20th century, public concern, scientific research, policies, and concerted management action were directed at such problems as overharvesting of fisheries (particularly oysters), infectious wastes, industrial and municipal pollution, toxic pesticides, wetland loss, channel dredging and spoil disposal, and power plant effects (Davidson et al., 1997). Except for fisheries, these were relatively localized problems. It was not until the last quarter of the century that it came to be appreciated that eutrophication had degraded the entire Bay ecosystem and had profound consequences to the Bay's resources (Malone et al., 1993). This is not surprising because awareness of the scope of marine eutrophication around the world (as evidenced by a 10-fold increase in scientific publications) was then beginning to emerge (Nixon, 1995). During the intervening period, however, reducing eutrophication has been the top priority for policy-making and management of the Chesapeake Bay.

In this paper we assess how eutrophication came to be recognized as an important problem; what we understand about it; commitments made to restore the Chesapeake ecosystem by reducing nutrient inputs; and the progress made in the restoration effort. We conclude by framing some challenges, particularly as related to reductions of agricultural nonpoint sources of nutrients—reductions that have been especially difficult to achieve. Our objective is to provide an overview of our scientific understanding and the scientific contributions toward restoring this eutrophic ecosystem. We hope, thereby, to provide a comparative reference for those addressing coastal eutrophication elsewhere in the world.

HYPOXIA AND EUTROPHICATION

Awareness

Although eutrophication became appreciated as a phenomenon affecting the entire Bay ecosystem only during the last quarter century, the more localized effects of organic enrichment have been addressed over a much longer period of time. The discharge of untreated or poorly treated wastes into harbors and tidal rivers, such as the Patapsco River–Baltimore Harbor, not only caused public health problems but also resulted in putrid conditions and algal blooms (Davison et al.,

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Abbreviations: BMP, Best Management Practice; NMP, Nutrient Management Plan; SWCP, Soil and Water Conservation Plan; WWTP, wastewater treatment plants.

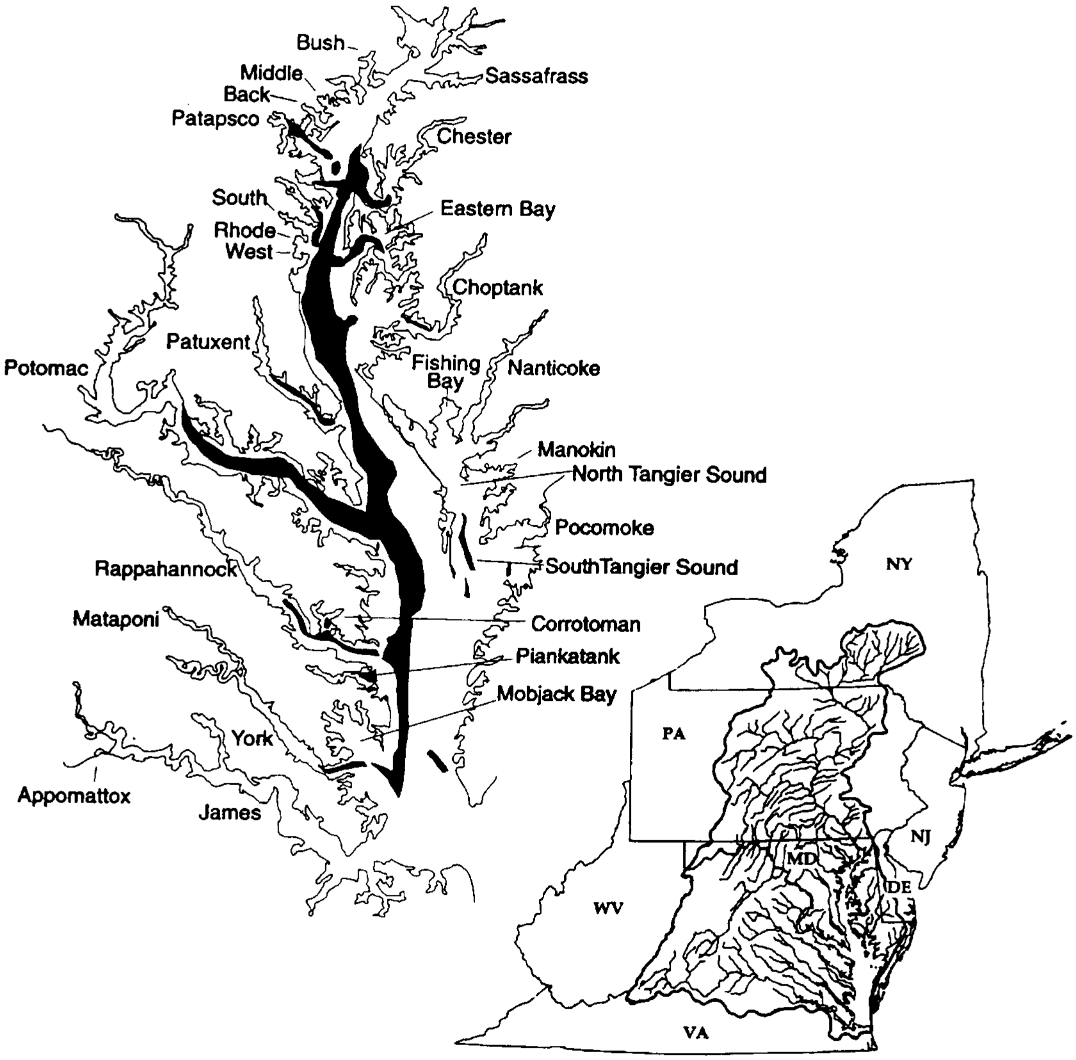

Fig. 1. The Chesapeake Bay and its major tributary subestuaries, showing the extent of hypoxic bottom waters (dissolved oxygen <2 mg L^{-1}) during the summers of 1994–1996 (Chesapeake Bay Program, 1997) and place names mentioned in this paper.

1997). By the 1960s the tidal freshwater reaches of the Potomac River subestuary below Washington, DC had become obviously overenriched as evidenced by massive blooms of cyanobacteria and macroalgae and depleted oxygen (Jaworski, 1990). Following the recent successes in addressing over-enrichment problems in Lake Erie and other lakes, major investments were made in 1972 to provide advanced waste treatment of metropolitan Washington, DC wastewaters, including phosphorus removal. Water quality dramatically improved, nuisance algal blooms retreated, and fish returned to the upper Potomac (Jaworski, 1990).

Research on the causes and effects of eutrophication expanded greatly following the Potomac experience, stimulated by growing concern about declining resources, evidence of worsening water quality in the Patuxent River and other tidal tributaries, the pervasive effects of Tropical Storm Agnes (also in 1972), and dramatic reductions in the extent of submersed aquatic vegetation throughout the Bay. The developing scientific understanding has driven and shaped environmental management not only in the Chesapeake, but also elsewhere in the world (Malone et al., 1993; Boesch, 1996).

Trends in Hypoxia

The most severe consequence of eutrophication is the depletion of dissolved oxygen by the decomposition of organic matter, either added to the ecosystem or produced within the ecosystem as a result of the stimulating effects of nutrient inputs. Anoxia (lack of oxygen) or hypoxia (dissolved oxygen concentrations lower than required by indigenous organisms) is a particular concern in coastal marine and freshwater bodies that exhibit density stratification permanently, seasonally, or periodically. Organic matter produced in lighted surface waters sinks to bottom waters where it decomposes, consuming oxygen inventories that are not replenished by photosynthesis or mixing with oxygen-rich surface waters. Hypoxic bottom waters have expanded during the latter 20th century in many coastal ecosystems influenced by land-based pollution (Diaz and Rosenberg, 1995; Boesch and Brinsfield, 2000), including large areas

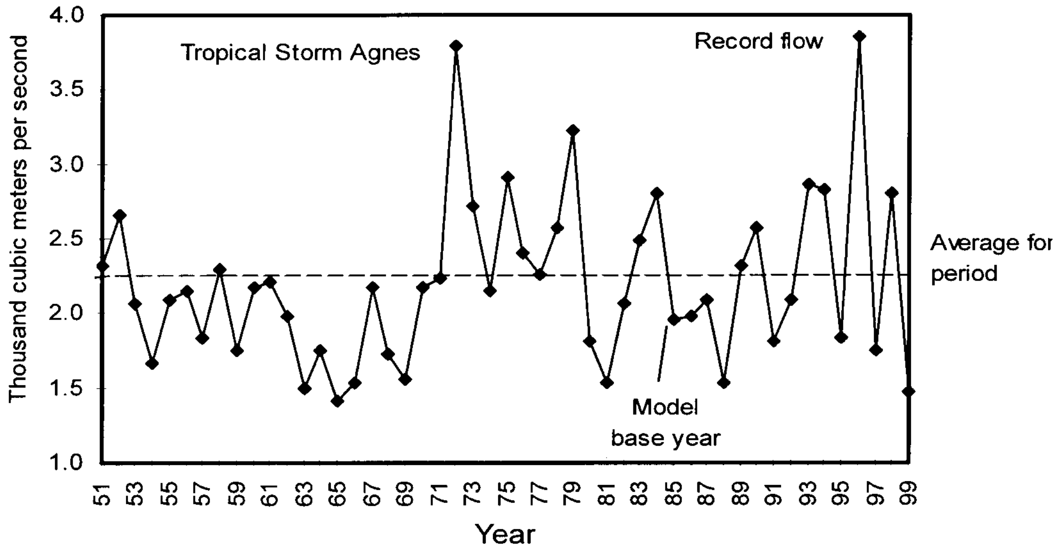

Fig. 2. Annual averages of the total freshwater streamflow entering the Chesapeake Bay from 1951 through 1999; high inflow during 1972 resulting from Tropical Storm Agnes; record annual flow year of 1996 resulting from both winter and summer floods; and 1985, the base year for nutrient reduction goals (U.S. Geological Survey data).

of the continental shelf of the northern Gulf of Mexico off the Mississippi River (Rabalais et al., 1996).

With a large catchment area within its watershed in comparison with its volume, its seasonally stratified water mass, and its isolated basins, the Chesapeake Bay is naturally susceptible to the development of hypoxic conditions. Seasonal hypoxia in deep waters of the central Bay was first reported from the mid 1930s (Newcombe and Horne, 1938). However, the degree to which hypoxia has worsened due to cultural (human) eutrophication has been the subject of scientific controversy.

In the early 1980s, debate as to whether hypoxia had become more extensive since the 1950s intensified. Taft et al. (1980) and Officer et al. (1984) presented comparisons that suggested a secular trend of worsening hypoxia. Seliger et al. (1985) argued that this apparent trend in fact represented variations due to climatic effects, particularly year-to-year variations in freshwater inflow (Fig. 2). The increase in hypoxia during the 1970s coincided with a transition from drought years of the 1960s to the higher-than-normal flow years of the 1970s. The extensive hypoxia witnessed in 1984 coincided with a high flow year. Greater freshwater inflows have the effect not only of increasing the loadings of nutrients from the watershed, but also of intensifying density stratification between fresher surface waters and saltier bottom waters.

Oxygen concentrations typically display high variability due to climatic, physical, and biological factors. Interpretation of long-term trends in oxygen concentrations is, consequently, frequently controversial (for example, see the debate between Gray and Abdullah [1996] and Johannessen and Dahl [1996] concerning historical changes in dissolved oxygen along the Norwegian coast). Long and reliable data records and methods to correct for climatic influences are ultimately required to resolve secular trends from transient effects. With the benefit of a longer record, a nonlinear relationship between the volume of hypoxic water in the Bay during the summer and freshwater inflow during spring was demonstrated (de Jonge et al., 1995). From the 1950s through the mid 1980s, hypoxic volume varied only slightly over years with low to medium flow, but was dramatically greater (as much as three times greater) during high flow years. However, W. Boicourt (personal communication, 1998) has found that the hypoxic volume observed during the eight years from 1985 through 1992 was two or more times greater for a given discharge rate than for the period 1949 to 1984. This suggests that some thresholds of nutrient loading or ecosystem response were reached, intensifying the hypoxia experienced for a given volume of freshwater discharged into the estuary. Since 1985, the volume of hypoxic water has remained high and less variable among years of relatively high or low flow.

Geochemical and biological indicators preserved in undisturbed sediments deposited in the hypoxic deep channel of the Bay have allowed the construction of a chronology of eutrophication and hypoxia in the Chesapeake. Sedimentation increased greatly following extensive land clearing and cultivation for agriculture beginning in the late 18th century and extending into the 19th century (Cooper and Brush, 1991). Increased deposition of organic carbon and biogenic silica (resulting from increased production of diatoms) reflect the growing enrichment of the estuary by nutrients following this landscape change. Even greater deposition of organic carbon (Cornwell et al., 1996) and lipid indicators of phytoplankton and bacterial production (Zimmerman and Canuel, 2000) occurred during the last half of the 20th century. A reduction of the diversity of the preserved diatom community and an increase in the ratio of centric to pennate diatoms reflect a shift from a benthic-dominated to a plankton-dominated, light-limited system that began nearly two centuries ago and became more dramatic during the last 50 years (Cooper, 1995). Seasonal anoxia intensified during the 1950–1980 time period, as reflected in the degree of pyritization of iron (Cooper and Brush, 1991), the ratio of acid volatile to chromium reducible sulfur (Zimmerman and Canuel, 2000), and assemblages of benthic foraminifera (Karlsen et al., 2000).

As a longer time record of hypoxia and other manifestations of eutrophication developed and as understanding of the effects of climatic variation on these properties increased, evidence has accumulated to support the case made by the sediment record for accelerated eutrophication during the last half of the 20th century. For example, Harding and Perry (1997) applied a statistical model to demonstrate a 5- to 10-fold increase in surface chlorophyll concentrations in the lower estuary from the early 1950s and a 1.5- to 2-fold increase elsewhere in the Bay. The increase in plankton biomass in the lower Bay cannot be accounted for by variability of freshwater flow and attendant properties (Harding, 1994; Harding and Perry, 1997). While it is clear that the Bay ecosystem had first become altered by enrichment 200 or more years ago, there can be little doubt that hypoxia and other consequences of eutrophication, including increased phytoplankton biomass, decreased water clarity, and loss of seagrass cover (Boynton, 1998), intensified greatly between the mid-1950s and mid-1980s. This was a period during which the human population in the Bay watershed nearly doubled (Davidson et al., 1997) and the use of inorganic fertilizers nearly tripled (Cornwell et al., 1996).

Role of Nutrients

Although there was earlier speculation about the effects of nutrient enrichment on oxygen depletion in the open Bay, it was probably the dramatic decline of submersed aquatic vegetation in shallow-water habitats during the early 1970s (Orth and Moore, 1983) that most stimulated actions to reduce nutrient inputs. Tropical Storm Agnes in 1972 had a great effect on the expansive beds of eelgrass (*Zostera marina* L.) and less salt-tolerant vascular plant species. The slow recovery of these grass beds led to allegations that the plants were succumbing to toxicity by herbicides such as atrazine, which came into widespread agricultural use in the late 1960s. Manipulative experiments coupled with field observations demonstrated that submersed vegetation was being stressed and killed by light limitation resulting from nutrient-stimulated growth of phytoplankton and epiphytes (Kemp et al., 1983; Twilley et al., 1985). Subsequent research has refined this understanding (Madden and Kemp, 1996) and allowed the development of ambient nutrient concentration goals for restoration of submersed vegetation (Dennison et al., 1993). Documentation of the effects of nutrients on submersed vascular plants in the Chesapeake Bay has since contributed to recognition of eutrophication as a key factor in the worldwide losses of seagrasses (Duarte, 1995).

Much has been learned about effects of different nutrients in stimulating plankton production in the estuary. Based on the paradigm of P limitation in freshwater ecosystems and the success in restoration of the tidal freshwater Potomac estuary following reductions in P loading (Jaworski, 1990), federal and state agencies sought during the 1970s to focus on P removal from sewage (Malone et al., 1993). Phosphorus inputs were thought to be dominated by point sources and more easily controllable. Moreover, the high ratio of N to P inputs to the Bay suggested that P limited phytoplankton production. Based on new perspectives that suggested that N, not P, was the primary nutrient limiting algal production in most marine ecosystems (Ryther and Dunstan, 1971), some Chesapeake Bay scientists argued that N sources should be controlled as well.

This controversy stimulated extensive research on factors limiting phytoplankton production in the Bay. Mesocosm experiments (D'Elia et al., 1986) and field bioassays involving the addition of combinations of nutrients (Fisher et al., 1992) have demonstrated that either P or N may limit production, with P being more limiting in lower salinity and N being more limiting in higher salinity and during the summer. Actually, the interrelationship of P and N stimulation is complex and must be considered over both spatial and temporal scales (Malone et al., 1996). Phytoplankton growth rates are limited by dissolved inorganic phosphorus (DIP) during the spring when biomass reaches its annual maximum and by dissolved inorganic nitrogen (DIN) during the summer when phytoplankton growth rates are highest (Magnien et al., 1992). Despite high inputs of DIN and dissolved silica (DSi) relative to DIP, seasonal accumulations of phytoplankton biomass within the hypoxic middle reaches of the Bay and the lower Bay are limited by riverine DIN supply. The magnitude of the spring diatom bloom is governed by DSi supply (Malone et al., 1996). Nearly two decades later, this synthesis underscores the importance of controlling both N and P inputs.

In addition to understanding the role of nutrients in stimulating organic production within the estuary, it is important to understand the nutrient sources, budgets, and trends in loadings. Boynton et al. (1995) undertook an ambitious synthesis of information on the inputs, transformations, and transport of N and P in the Chesapeake Bay. Total N and P inputs from land and the atmosphere were estimated to represent 6- to 8-fold and 13- to 24-fold increases, respectively, in loads to these systems from pre-colonial times to the mid 1980s. This is consistent with larger-scale estimates of Howarth et al. (1996). Approximately one-fourth of the N and one-third of the P are from point sources (Table 1) and most of the rest is from diffuse sources on the landscape and from the atmosphere. Atmospheric deposition of N is significant, with direct deposition onto tidal waters ac-

Table 1. Comparison of estimates of sources of nitrogen and phosphorus entering the Chesapeake Bay from land and the atmosphere. Values are percent of total loads.

	Boynton et al. (1995)		Magnien et al. (1995)	
Nutrient source	Nitrogen	Phosphorus	Nitrogen	Phosphorus
	%			
Point sources	28	35	23	34
Diffuse sources	60	58	66	60
Agriculture			39	49
Forests			18	3
Urban/suburban			9	8
Atmosphere (direct)	12	7	11	6

counting for 11 to 12% and export of N deposited on forests and other land types probably accounting for a like amount. Boynton et al. (1995) attempted to balance the nutrient budgets through estimates of burial, denitrification (loss of N_2 to the atmosphere), fisheries harvest, and losses to the ocean. They reached the surprising conclusion that while 30% of the N inputs are eventually lost to the ocean, the Chesapeake Bay actually is a significant net importer of P from the ocean.

Effects

Given the level of concern about the eutrophication of Chesapeake Bay and the public and political commitment to nutrient reduction, there is surprisingly little scientific documentation of the effects of hypoxia and eutrophication on living resources. Seasonal hypoxia certainly reduces the abundance, diversity, and productivity of benthic animals in the affected deep-water habitats (Holland et al., 1987; Diaz and Rosenberg, 1995). Expanding hypoxia has also affected some deeper oyster reefs, although the great reductions in oyster populations witnessed this century are due primarily to overharvesting and effective "mining" of the reefs (Rothschild et al., 1994). On the other hand, the decline in filter feeding by the once abundant oyster populations is thought to have reduced the resilience of the Bay ecosystem to eutrophication (Baird and Ulanowicz, 1989).

Presumably, bottom-dwelling and feeding fishes have also been disadvantaged, as is typical where eutrophication results in seasonal hypoxia (Caddy, 1993). Catches of some bottom-feeding fishes have declined, but effects of overharvesting complicate the linkage with hypoxia. In addition, much of the concern about the loss of submersed aquatic vegetation due to eutrophication is based on its value as habitat for juvenile fish and crustaceans. Reduction of this habitat has been suggested as a factor in declines in populations of blue crabs, the Bay's most valuable fishery (Pile et al., 1996). However, fishing mortality and environmental factors affecting year-to-year variations in recruitment cloud the influences of habitat loss. Most of the fishery productivity of the Chesapeake Bay now consists of species dependent on planktonic food chains, such as menhaden, which may have become more productive as a result of nutrient enrichment (Houde and Rutherford, 1993).

The Chesapeake Bay is particularly important as a spawning and nursery ground for species such as striped bass, whose nursery habitat is susceptible to hypoxia and alterations of food chains due to eutrophication. Of course, both the life histories of fish and the food chains supporting their productivity are complex. Dissolved oxygen conditions not low enough to kill fish larvae may increase their susceptibility to predators and decrease their ability to capture prey (Breitburg et al., 1997). Moreover, several lines of evidence and general theory suggest that eutrophication in the Chesapeake has altered trophic networks by shortening food chains, increasing microbial production, and decreasing the proportion of metazoan production (Baird and Ulanowicz, 1989). Simple analysis of the effects of eutrophication limited to the suitability of oxygen concentrations for survival of adult animals underestimate these effects.

New understanding of the flow of energy and materials in the Chesapeake ecosystem has provided explanations for why the Bay is highly productive but particularly susceptible to dysfunction from eutrophication. Compared with other marine ecosystems, the Chesapeake has higher primary production than would be predicted from known nutrient inputs (Nixon et al., 1986; de Jonge et al., 1995). This is because of its size, material residence times, and tidal and nontidal circulation, which lead to a greater recycling and reuse of nutrients.

Compared with other coastal ecosystems in which about one-half of the N inputs are removed by the microbial processes of denitrification, only about one-fourth of the N entering the Chesapeake is lost due to denitrification (de Jonge et al., 1995; Nixon et al., 1996). This lower denitrification efficiency is related to the anoxic conditions overlying a large portion of the Bay's bottom sediments during the summer months (Kemp and Boynton, 1992; Fig. 3). Lack of oxygen limits nitrification (the aerobic microbial processes by which ammonium produced by organic decomposition is converted to nitrate), a precursor of denitrification (an anaerobic metabolic process converting nitrate to N_2 gas). Simply put, denitrification is limited by availability of nitrate. At the same time, phosphate flux out of anaerobic sediments increases because sulfide out-competes phosphate for iron binding sites. These positive feedbacks result in rapid recycling of P and attenuation of an important N sink.

In a degradation trajectory these feedbacks result in increased phytoplankton production, decreased water clarity, and increased oxygen demand in bottom waters. Under a restoration trajectory, this may actually turn out to be good news wherein even modest increases in bottom water dissolved oxygen (say from 0 to 1 mg L^{-1}) may have surprising and profound beneficial effects (D'Elia et al., 1992).

Nutrient Dynamics in the Watershed

In contrast to the 25 years of intense research on dynamics of nutrients and primary production in the estuary, the trends and dynamics of nutrients in the Chesapeake Bay watershed have been relatively little studied. This is unfortunate because now that the scope of estuarine eutrophication has motivated commitments to reduce nutrient inputs in order to restore the ecosystem, reductions of these sources must rely heavily on minimizing losses and maximizing sinks within the watershed. For management purposes nutrient delivery and loss rates are assumed for various land covers and uses in a Chesapeake watershed model that predicts nutrient inputs to the Bay (Donigian et al., 1994). As will be discussed later, this includes crediting nutrient reductions for the application of various land management practices. The assumed reductions are propagated downstream.

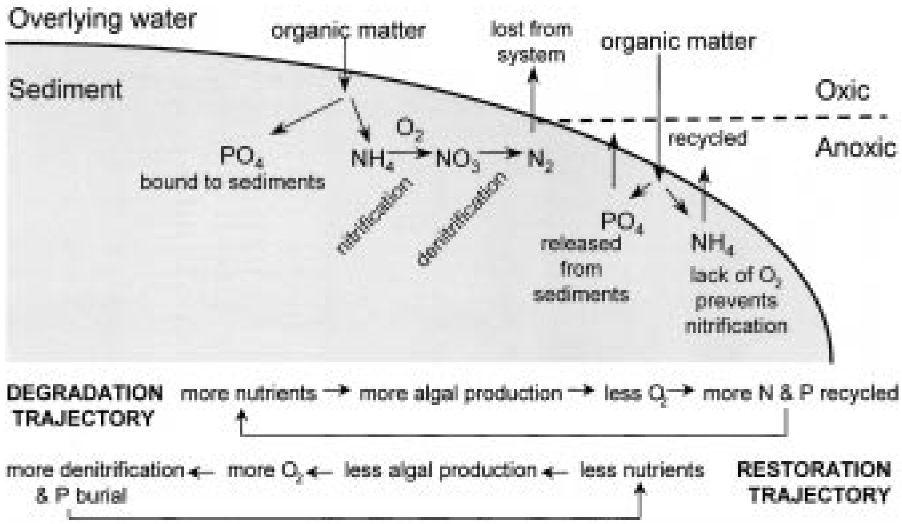

Fig. 3. Bottom sediments play an important role in recycling nutrients and as nutrient sinks, depending on the availability of dissolved oxygen. Positive feedbacks worsen eutrophication as coastal ecosystems degrade because more nutrients are recycled to the water column under anoxic conditions. The feedbacks work in the opposite direction as ecosystems are restored, by reducing nutrient loadings as more phosphorus is buried and more nitrogen is lost to the atmosphere.

Nutrient exports from small watersheds vary greatly depending not only on land use and management practices, but also on physiography, vegetation cover, and the underlying geology, which affects water chemistry and ground water transport and residence time. The extent and characteristics of riparian transition zones between landscapes and streams also play important roles (Correll et al., 1992; Jordan et al., 1993). For example, in a small watershed on the Bay's western shore, riparian deciduous hardwood forest bordering croplands removed more than 80% of the nitrate and phosphate in overland flows and about 85% of the nitrate in shallow ground water drainage. Nonetheless, the small amounts of N and P that discharged into tidal waters were large enough to cause overenriched conditions in the tidal creeks.

Phosphorus and nitrogen behave quite differently within the watershed. Phosphorus tends to be particle-bound, so its transport is dependent on soil type, slope, rainfall intensity, and particle trapping capabilities of riparian zones, wetlands, and reservoirs. Runoff of dissolved P can nonetheless occur (Staver and Brinsfield, 1995b; Coale, 1999). Nitrogen is transformed into highly soluble nitrate, tends to leach from soils into ground water, and is subject to losses due to denitrification. In agricultural watersheds on the Delmarva Peninsula, N discharges increase as the proportion of cropland in the watershed increases, while P discharges do not correlate as well with land use but are influenced more by transport of suspended particles (Jordan et al., 1997).

Forests cover approximately 58% of the Chesapeake Basin, yet release less than 18% of the N reaching the Bay. While forests have historically acted as an N sink, deposition and retention of atmospherically deposited N vary widely among forests within the basin, depending on their location, age, and degree of disturbance by humans and herbivores. Retention factors range from 28 to 98% (Gardner et al., 1996). Furthermore, evidence is increasing that excess nitrate deposition may be acidifying soils and otherwise reducing the degree to which forests are able to retain N.

Total input and output budgets for large watershed areas (Howarth et al., 1996; Jordan and Weller, 1996), the generalized watershed models used in Chesapeake Bay Program (Donigian et al., 1994), and spatially referenced regression modeling based on stream monitoring (Preston and Brakebill, 1999) are useful as tools for identifying the relative importance of sources for strategic targeting. However, land uses are changed and management practices are applied on a hectare-by-hectare basis. It is becoming increasingly useful to develop and apply spatially explicit watershed models that take into account more realistic hydrological behavior and landscape grain to predict nutrient delivery (National Research Council, 1994).

EVOLUTION OF PUBLIC POLICY

Toward the 1987 Agreement

Malone et al. (1993) examined in considerable detail how views about the importance of nutrient loadings to the Chesapeake Bay began to change among scientists, managers, and policy makers. The recovery of the tidal Potomac following improved wastewater treatment at Washington, DC, in 1972 had the effect of instilling confidence in regional environmental managers that commitment to waste treatment would yield positive results. However, this success also had the effect of focusing attention on point sources of pollutants, obscuring the role and effect of nonpoint sources on the estuary.

Also in 1972, record floods associated with Tropical Storm Agnes affected the entire Chesapeake watershed. The resulting freshet had profound effects on the Bay and its tidal rivers. It forced both the scientific and the management communities to begin to think of the Bay not as a vast arm of the sea, but more as an estuarine river mouth heavily influenced by land use in the watershed. Following Agnes, concern about the large-scale changes in the Bay, particularly the declines in submersed aquatic vegetation, provoked Congressional pressure for a better understanding how these stresses could be alleviated. The resulting multiyear Chesapeake Bay Study began in 1978 under the USEPA and focused on aquatic vegetation, toxic substances, and nutrient enrichment. Workshops and conferences dealing with nutrient enrichment helped to coalesce opinions about the importance of nonpoint sources, but as was discussed earlier, it was the studies of submersed vegetation that truly raised eutrophication to the top of the list of Bay problems.

At the same time, controversies developed over the effects of population growth and expanding wastewater discharges into the Patuxent River. Estuarine scientists who had been studying the brackish Patuxent had become alarmed at signs of over-enrichment and, in particular, suspected N enrichment. State and federal environmental officials and their engineering consultants, borrowing from the experience of the tidal freshwater Potomac, were planning waste treatment facilities with P- but without N-removal capabilities. Officials from the rural counties around the lower river sued federal and state agencies to require N removal, with estuarine scientists appearing for the plaintiff in opposition to the very agencies that supported their research. The matter was ultimately settled in 1981 by a "charette" in which the parties committed to reach a consensus during a time-constrained meeting. The agreement to remove N is considered a milestone in Bay management. Interestingly, the evidence supplied by the scientists was largely inferential; the extensive experiments and observations earlier reviewed later confirmed the wisdom of a decision made on limited scientific evidence.

With the conclusion of the five-year Chesapeake Bay Study, the three states (Virginia, Maryland, and Pennsylvania) that occupy most of the Bay's watershed, the District of Columbia, and the federal government represented by the Environmental Protection Agency endorsed the first Chesapeake Bay Agreement in 1983. A scientific and technical advisory committee was formed and it released a report in 1986 that presented evidence that both P and N removal would be required to improve water quality in the Bay and its tributaries. Very importantly, the report also identified cost effective technologies available for the combined removal of these nutrients from point-source discharges. This scientific consensus provided the rationale and credibility for the bold action of the Second Chesapeake Bay Agreement in 1987. This agreement committed the signatories to implementation of "a basin-wide strategy to equitably achieve by the year 2000 at least a 40% reduction of nitrogen and phosphorus entering the main stem of the Chesapeake Bay." It indicated that this "strategy should be based on agreed upon 1985 point source loads and on nonpoint loads in an average year."

The 40% reduction goal was reached based on modeling available at the time coupled with subjective judgment of the levels required to return water quality to conditions that existed in the 1950s (Malone et al., 1996). Although the 1987 Agreement did not make such a differentiation, the commitment was interpreted as a 40% reduction on controllable sources of N and P. Nutrient inputs from atmospheric sources, from watershed states not party to the Agreement (New York, Delaware, and West Virginia), and from background inputs estimated for a forested watershed were excluded from the determination of the controllable loads. Thus, the commitment translates to 24 and 35% reductions of total average loads of N and P, respectively, to the Bay. This was defined as 74 million pounds (33.6 million kg) of N and 8.4 million pounds (3.8 million kg) of P on an annual basis in the Executive Council's 1993 Joint Tributary Strategy directive.

The Chesapeake Bay Program

Although at times a frustrating experience to the individuals involved, the evolution of this landmark policy for nutrient load reduction actually proceeded quickly in a relative sense. The development and refinement of Chesapeake Bay Program objectives, coupled with the Program's use of monitoring and assimilation of new scientific information, have been cited as an effective application of adaptive management (Hennessey, 1994). However, those of us who toil daily at the science–management interface wish for the greater emphasis on the learning and experimentation, tighter linkages, and shorter time steps envisioned in the adaptive management ideal (Boesch, 1996).

A decade has passed since the historic 1987 Agreement and the Chesapeake Bay Program has persistently pursued the nutrient reduction goals. It has maintained the commitment and involvement of the signatory parties, despite many changes in political leadership in the jurisdictions. In fact, the personal engagement of the three governors, the mayor of the District of Columbia, the administrator of the Environmental Protection Agency, and the representative of the state legislators who together constitute the Chesapeake Bay Executive Council has been critical. They meet on an annual basis and, prior to the year 2000, issued no fewer than 24 additional directives, agreements, and amendments that advance the goal of reducing eutrophication, as well as commitments to reduce toxic substances, restore habitats, manage shared fisheries, and pursue other objectives.

Noteworthy among these various agreements and directives are the following related to reducing eutrophication:

- The development of tributary-specific nutrient reduction strategies that take into account localized environmental quality goals as well as the baywide 40% reduction goal and involve local stakeholders.
- The adoption of living resource restoration of as the overarching goal, with specific numerical objectives for submersed aquatic vegetation beds recovered as a key living resource indicator.
- The adoption of the goal of restoring riparian forest

Table 2. Commitments of the Chesapeake 2000 Agreement related to eutrophication.

Category	Commitment
Living Resource Protection and Restoration	• Achieve a 10-fold increase in native oysters (this is intended to improve water clarity by increasing biofiltration).
Vital Habitat Protection and Restoration	• Recommit to goal of restoring 114 000 acres (46 170 ha) of submerged aquatic vegetation and by 2002 revise restoration goals and strategies to reflect historic abundance; revised goals will include specific levels of water clarity to be met in 2010. • By 2010 develop and implement locally supported watershed management plans in two-thirds of the Bay watershed; these plans would address the protection, conservation, and restoration of stream corridors, riparian forest buffers, and wetlands. • By 2010 achieve a net gain of 25 000 acres (10 125 ha) of tidal and nontidal wetlands. • Ensure that measures are in place to meet riparian forest buffer restoration goal of 2010 miles (3234 km) by 2010.
Water Quality Protection and Restoration	• Continue efforts to achieve and maintain 40% nutrient reduction goal agreed to in 1987. • By 2010 correct nutrient- and sediment-related problems in the tidal waters sufficient to remove them from the list of impaired waters under the Clean Water Act; by 2001 define conditions necessary to protect living resources and assign N and P load reductions; by 2002 revise Tributary Strategies; and by 2003 adopt new or revised water quality standards. • By 2003 assess the effects of airborne nitrogen compounds on the Bay ecosystem and help establish reductions goals.
Sound Land Use	• Permanently preserve from development 20% of the land area in the watershed by 2010. • By 2012 reduce the rate of harmful sprawl development of forest and agricultural land in the watershed by 30%. • By 2002 promote coordination of transportation and land use planning to minimize adverse effects on the Bay.
Stewardship and Community Engagement	• Promote individual stewardship and assist individuals and community-based organizations, businesses, local governments, and schools to achieve goals of Agreement.

buffers along 2010 miles (3234 km) of stream and shoreline in the watershed by the year 2010.
- The conduct of two major mid-course reevaluations (in 1991 and 1997) of progress toward the 40% nutrient reduction goal (discussed in the next section).

A comprehensive new agreement was signed by the Executive Council in June 2000 (Table 2). It recommits to the 40% nutrient reduction goal to be achieved through Tributary Strategies, but prescribes a process aimed at removing the Bay and its tributaries from the list of impaired waterbodies under the Clean Water Act. This will probably require more substantial reductions in nutrient loadings for some parts of the estuary. Total maximum daily loads (TMDLs) needed to achieve water quality standards (focusing initially on dissolved oxygen, chlorophyll concentrations, and water clarity) will be determined and implemented by the state jurisdictions. In addition, a number of new or reformulated commitments were made concerning living resources, habitats, land use, and stewardship that are intended to contribute to a reversal of eutrophication.

Of course, most of the work of the Chesapeake Bay Program takes place outside of the annual meetings of the Executive Council through the extensive network of committees and subcommittees that engage many hundreds of individuals. The Implementation Committee is comprised primarily of state and federal agency representatives responsible for implementing the program's policies. It has a number of subcommittees dealing with such issues as nutrients, living resources, monitoring, and modeling. The Scientific and Technical Advisory Committee, comprised mainly of academic and federal scientists, the Citizens Advisory Committee, and the Local Government Advisory Committee are separate, standing committees advising the program. Underpinning this regional program structure are the numerous management, enforcement, assessment, and assistance programs of the state and federal agencies themselves, a variety of nongovernmental organizations that involve citizens at the grassroots and political levels, and the activities of the regional scientific community.

PROGRESS IN RESTORATION

Actions to Reduce Nutrients

As previously discussed, numerical modeling of nutrient fluxes through the entire Chesapeake Bay basin has been used to define specific nutrient loading goals in units of mass for various areas of the watershed. This allowed the 40% nutrient reduction goals for controllable nitrogen and phosphorus to be apportioned uniformly among the states and major watersheds in the region. Furthermore, these goals were considered loading *caps*, with the intention that nutrient loadings would be kept below the caps in perpetuity.

Following the 1987 Agreement, planning for the implementation of various point- and nonpoint-source nutrient controls was intensified. By 1987, phosphorus reductions at wastewater treatment plants (WWTPs) were well underway, in many cases due to decisions predating the 1987 Agreement. Because the 1987 Agreement benchmarked nutrient loads to 1985, many of these load reductions were counted toward fulfilling the 40% reduction goals. Technology for P removal from the WWTP effluent was well understood and reliable. At this time, bans on phosphate-based detergents were also going into effect in all of the Bay watershed states, significantly reducing both influent and effluent P concentrations in the wastewater stream.

Wastewater treatment for N was not as well understood and feasibility studies were conducted to determine the ability to implement N removal from large wastewater volumes. One of the earliest and boldest moves toward nitrogen reduction was the decision following the 1981 charette to have all of the major WWTPs in the Patuxent River drainage implement nitrogen removal technology by the early 1990s. The breakthrough technology for N removal in the Chesapeake Bay basin has been termed *biological nutrient* *removal*, or BNR (Randall et al., 1990). Biological nutrient removal relies on nitrification followed by denitrification—the same processes that contribute to the atmospheric sink in the natural ecosystem (Fig. 3)—to reduce nitrogen concentrations in the effluent. As of 1997, 33 (mostly in Maryland) of 315 major WWTPs (those with effluent flows >19 million liters per day) in the signatory states had BNR operational (providing treatment to 26% of the effluent load) and 57 had plans to implement this technology by the year 2000 or shortly thereafter. This will result in BNR treatment of approximately 58% of the wastewater flow to the Bay from the signatory jurisdictions.

Less progress has been made in implementing nutrient controls from nonpoint sources. The 1997 reevaluation estimated that point sources of N and P had been reduced by 15 and 58%, respectively, between 1985 and 1996, while nonpoint sources were estimated to have been reduced by only 7 and 9% and were projected to reach 19 and 15% by 2000 (Chesapeake Bay Program, 1997). In the more developed areas of the basin, accounting for about 11% of the watershed area, regulations have been applied since the mid-1980s to manage stormwater runoff and to control erosion during and after commercial and home construction. Nonpoint source controls are estimated to have reduced P losses from developed landscapes by more than half, but have had less effect on N losses. There are also a number of programs that target the protection and restoration of sensitive lands, such as those with highly erodable soils, wetlands, and riparian forests.

Significant efforts have also been made in implementing agricultural practices that reduce nutrient and sediment losses from the land. Agricultural lands account for about one-third of the watershed land area, but contribute an estimated 39 and 49% of the total N and P loads, respectively, that reach the Bay's tidal waters (Magnien et al., 1995). This makes agriculture the single largest source of nutrients to the Bay and its tributaries. Agricultural nutrient reduction practices include those that reduce surface runoff, prevent the application of nutrients in excess of crop nutrient uptake or retain excess nutrients in the soil during colder seasons. These are the subject of more detailed discussion under Reducing Agricultural Sources.

Model Projections

Targeting and assessing the progress of the nutrient reduction efforts of the Chesapeake Bay Program have relied heavily on numerical models that predict the sources and transport of nutrients through the watershed and in the estuary and resulting effects on Bay water quality. These include linked models of atmospheric transport and deposition, watershed land-cover and hydrologic transport, and three-dimensional time-variable hydrodynamics in the Bay. The hydrodynamic model is coupled with a three-dimensional, time-variable model of water quality, including sediment processes (Cerco, 2000). Calibration analyses of the entire modeling structure have been conducted using data collected during a three-year period from 1984–1986. Freshwater inflow was relatively high in 1984 and somewhat below average in 1985 and 1986 (Fig. 2). Together, these models are considered the state-of-the-art in estuarine water quality modeling.

In the 1991 reevaluation, the water quality model outputs under several reduced N and P loading scenarios yielded a number of interesting results (Cerco, 1995). The most important prediction was that anoxia in the Bay would not be eliminated with a 40% controllable N and P reduction—in fact that would take a nearly 90% reduction. Hypoxic volume-days, a measure of the extent and duration of oxygen levels below 1 mg L^{-1}, were estimated to be reduced by 20% from the base year by the 40% reduction. Additional reductions of atmospheric inputs resulting from implementation of Clean Air Act controls on emissions would increase this to 30%, close to what would be achieved by reducing controllable N and P inputs to what was assumed to be the limits of technology. The model also confirmed the particular importance of N reductions on summertime hypoxia. The model predicted that lower P loading reduces primary production in the upper Bay, but would result in less photosynthetic oxygen production and more transport of N down the Bay, where N is limiting. The water quality model will play a central role in determining the total maximum daily loads (TMDLs) for the Bay and its tributaries in executing the 2000 Agreement.

A revised version of the watershed model was used in the 1997 reevaluation to estimate nutrient load reductions achieved to date for the 10 major watersheds and to project load reductions expected to be achieved by the year 2000 (Chesapeake Bay Program, 1997). These projections were not linked with the estuarine water quality model as the latter was being revised. The watershed model projections estimate that the practices in place in 1997 had nearly achieved the P-load reduction goal, which should be exceeded by 2000 (Fig. 4). On the other hand, estimated N-load reductions have been more modest and were not expected to meet the year 2000 goal. More recent refinements of the watershed model suggest that the simulations in the 1997 reevaluation were overly optimistic and that the P-load reduction goal may have just been met and the N-load reduction goal missed by a larger margin by 2000. The differences are largely attributable to lower and more realistic assumptions about the effectiveness of agricultural management practices and to new estimates of loadings from animal wastes that are higher than previously assumed.

Because they yield clear numerical results with which to gauge progress, the models have a seductive appeal to policy makers and managers, an appeal that risks false confidence and misconception. It should be remembered that numerical models of such complex systems have more heuristic than deterministic value (Oreskes et al., 1994). Specifically, three caveats need to be appreciated in interpretations of the watershed–water quality models: (i) the model predictions are very sensitive to several uncertain assumptions, (ii) the models predict "average" conditions in a variable world, and

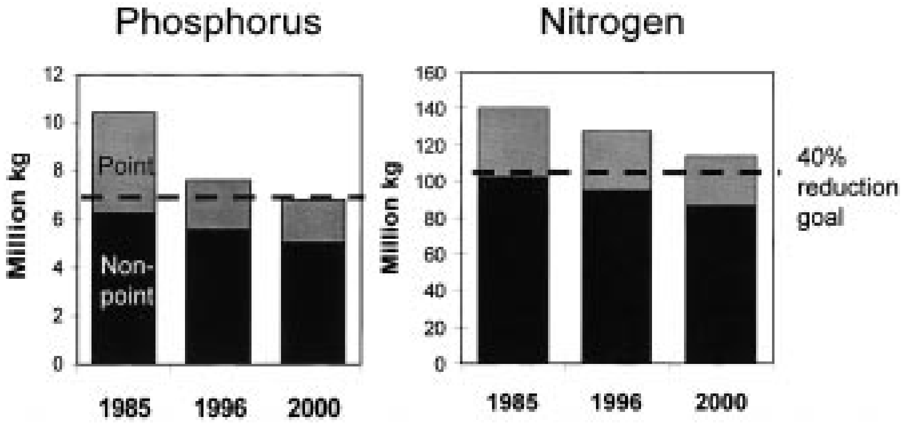

Fig. 4. Watershed model estimates of controllable loads of nitrogen and phosphorus into the Chesapeake Bay for 1985 (the base year), 1996 (based on source reductions put in place), and 2000 (the targeted year for the 40% reduction goal). For 1996 and 2000 these are estimates adjusted to the average freshwater flow year and not estimates of nutrients that actually have been or will be delivered (Chesapeake Bay Program, 1997).

(iii) the models assume immediate benefits of source reductions in the Bay's tidal waters, when in fact there may be significant lag times involved.

While some source reductions can fairly accurately be measured, for example through effluent monitoring of regulated point sources, nonpoint-source reductions are estimated based on "typical" losses for land use categories and assumed effectiveness of practices such as stormwater management, agricultural management practices, or riparian buffers. For example, if a farmer develops a nutrient management plan, the model credits formulaic N and P loading reductions for each particular practice in the plan. This assumes that the farmer actually implemented the plan and that, individually and collectively, the practices accomplish the loading reductions credited. Under Reducing Agricultural Sources, below, doubts are raised about these assumptions. Furthermore, there are numerous and uniformly applied assumptions about nutrient losses within the watershed that are based on limited field data from what is, after all, a heterogeneous world.

While assessing progress based on average conditions and immediate benefits may be reasonable for management purposes, this does not reflect actual ecosystem responses. Lag factors such as ground water transport pathways and temporary retention of nutrients in streambeds or reservoirs during transport to tidal waters cause delays between the implementation of nonpoint source controls and actual nutrient load reductions to the Bay's tidal waters. Interannual variations in precipitation also greatly affect the nutrient load that actually reaches the Bay and biogeochemical responses in the estuary (Boynton and Kemp, 2000). For example, during recent years of unusually high freshwater flow into the Bay (Fig. 2), nutrient loadings have actually increased relative to earlier years when there were fewer source controls in place. The increased delivery of nutrients during higher freshwater inflow poses an obvious management challenge under any climatic shift toward higher springtime precipitation and runoff, as both observed trends and regional climate models suggest (Giorgi et al., 1994; Karl et al., 1995; Justić et al., 1996; Najjar et al., 2000).

Monitoring Recovery

The Chesapeake Bay Program has had in place since 1985 a large and ambitious program for monitoring environmental conditions, water quality, and biota. Data collected in the tidal waters of the Bay and its tributaries over the last 14 years reveal several patterns that can be related to both implementation of nutrient reduction practices and natural environmental variations, particularly variations in freshwater flows. Declines in flow-adjusted concentrations of total N or P in stream flow have been observed from 1985 and 1998 in several major rivers as they discharge to tidal waters, including the Susquehanna, Patuxent, Rappahannock, and James (Langland et al., 2000). In the open waters of the mainstem Bay, there have been no statistically significant trends in nutrient concentrations (Chesapeake Bay Program, 1997). However, in areas of the Bay receiving high loadings from WWTPs, where significant and demonstrable load reductions have been achieved, nutrient concentrations have declined in tidal tributaries, especially for phosphorus. In the Patuxent River, previously mentioned as the site of implementation of N controls at all major WWTPs, N concentrations have also declined. In tidal tributaries dominated by nonpoint sources, nutrient concentrations have generally not declined and, in some areas, nutrient concentrations actually increased over the 12-year period. Statistical analyses to adjust for the effects of variations in freshwater flows reveal that at least some of these increases are due to the generally greater flows experienced during many of the years since 1992 (Fig. 2).

The Bay monitoring data do not reveal any significant changes in dissolved oxygen concentrations in areas of summer hypoxia of deep bottom waters (Fig. 5a). In these areas, typically the middle reaches of the mainstem and several major tributaries (Fig. 1), there generally have been either modest or no significant improvements noted in nutrient concentrations or algal biomass. These

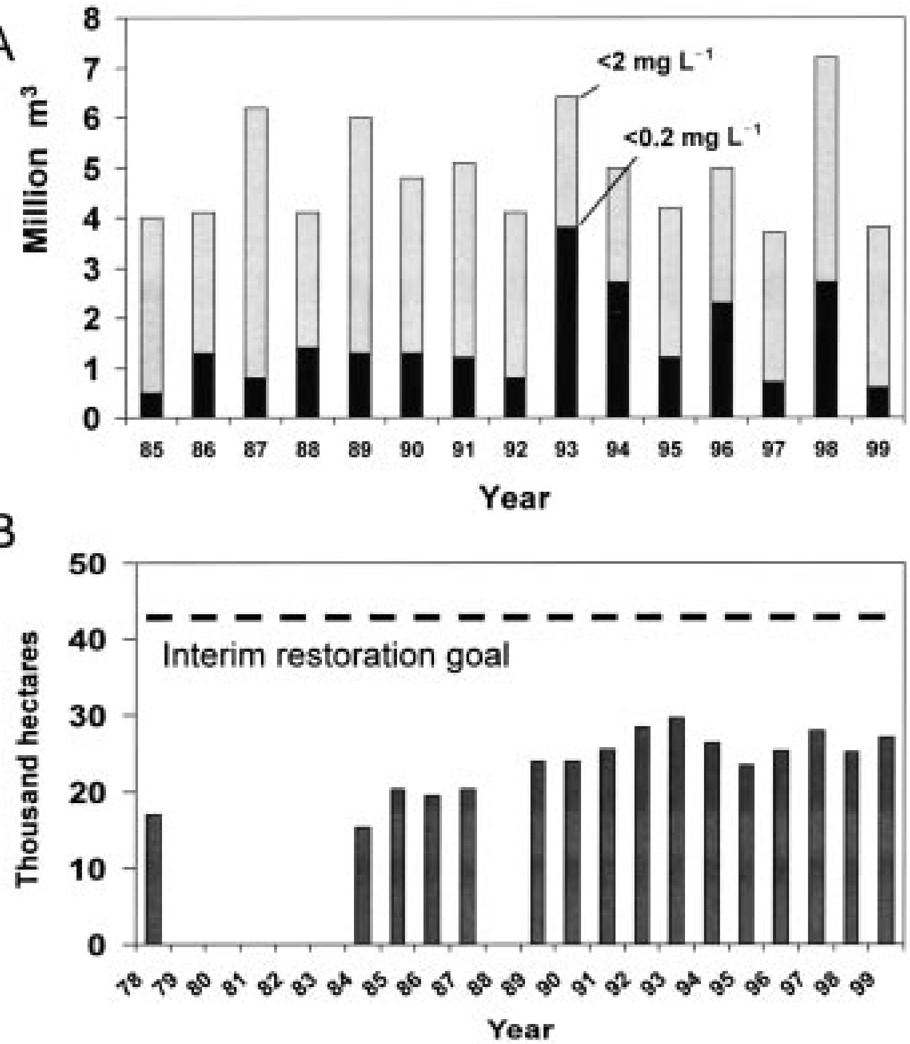

Fig. 5. (*A*) Volume of summer hypoxic water in the mainstem of the Chesapeake Bay. (*B*) Area of Chesapeake Bay bottom habitat covered by beds of submersed aquatic vegetation based on annual aerial surveys compared with the interim restoration goal.

are also areas more subject to N limitation of algal growth. Because progress in load reductions for N have generally lagged those for P, and recent high freshwater loads have delivered higher nonpoint source inputs of nutrients, it is reasonable to expect that several more years may be required to realize improvements in the Bay's hypoxia problems. On the other hand, there are some encouraging signs in the recovery of submersed aquatic vegetation in the Bay (Fig. 5b). While the area revegetated is yet far below the recovery goal set by the Chesapeake Bay Program, and reversals were noted during the recent high flow years, significant recovery has been noted from the 1984 low point.

REDUCING AGRICULTURAL SOURCES

Existing Efforts

Because agriculture is the largest source of both nitrogen and phosphorus entering the Chesapeake Bay (Table 1), it was recognized that major reductions in nutrient transport from agricultural areas would be necessary to achieve the nutrient-reduction goals. Efforts to reduce agricultural nonpoint sources during the past 12 years have focused mainly on education, technical assistance, and cost-share programs to promote the use of practices that reduce nutrient and sediment losses from agricultural land. These include Soil and Water Conservation Plans (SWCPs), Nutrient Management Plans (NMPs), and state agricultural cost-share programs that provide money to landowners for implementing practices prescribed in the plans. These practices, referred to as Best Management Practices (BMPs), are special techniques or structures that help prevent soil erosion, reduce unneeded nutrient application, and control nutrient movement. Soil and Water Conservation Plans include stream bank protection measures such as fencing and livestock crossings, animal waste storage structures, grassed waterways, and reduced tillage systems designed to control soil erosion. These practices focus primarily on improving surface drainage water quality by decreasing sediment transport. As a result they offer a means for reducing sediment-bound phosphorus and nitrogen, as well as other potential pollutants that bind readily to soil particles. Nutrient Management Plans integrate various BMPs and fertility recommendations on a field-by-field basis with an overall goal of matching nutrient requirements with realistic crop yield goals.

Through 1998, nutrient management plans had been developed for 770 000 ha of agricultural lands in the signatory states (26% of the total agricultural lands) in comparison with the year 2000 goal of 1.3 million ha. In Maryland, for example, more than 1280 new nutrient management plans were developed and certified in 1996, bringing the total amount of cropland in Maryland under nutrient management plans to more than 338 000 ha. In addition, plans affecting some 105 000 ha were updated to ensure their continued efficiency in managing crop nutrients following changes in farming operations (Maryland Department of Agriculture, 1996).

Effectiveness of Management Practices

Although strategies designed to reduce soil erosion have been successful in reducing phosphorus transport in areas of the Bay watershed predisposed to high erosion rates, these approaches are less effective in addressing dissolved nutrient transport in surface-water runoff or leaching to ground water. For example, in the Maryland Coastal Plain, which represents the most intensive row crop region in the watershed, ground water flow is the dominant hydrologic link between agricultural systems and surface waters.

Monitoring of continuous corn (*Zea mays* L.) production systems on the Delmarva Peninsula showed that annual ground water recharge volume is three times greater than surface runoff volume. Annual nitrate N losses to ground water are five times greater than total nitrogen losses in surface water runoff (Staver and Brinsfield, 1995a). In addition, the lag time (as much as 10 years) for movement of nitrate through ground water flow systems complicates efforts to clearly link potential reduction in nitrate inputs to cropping systems with changes in surface water quality.

The combination of grassed waterways, continuous no-till, splitting nitrogen application, and fertilizing for realistic yields resulted in long-term annual losses of total N in surface water runoff of <5 kg ha^{-1}. However, ground water nitrate N concentrations reached >10 mg L^{-1} and annual nitrate N leaching losses approximately 30 kg ha^{-1}, even when corn yield goals were met (Staver and Brinsfield, 1995a). Thus, achieving significant reduction in total nitrogen losses depends largely on effective strategies to reduce nitrate-leaching losses to ground water.

The problem is further exacerbated by soybean [*Glycine max* (L.) Merr.] production and the application of animal manure and sewage sludge to cropland. Recent studies show that nitrate leaching losses following soybean harvest can be equivalent to those from corn production with its heavy fertilizer requirements (Angle, 1990). Furthermore, the application of organic waste to corn production systems using an N-based NMP consistently results in elevated nitrate leaching losses compared with applying inorganic N at recommended rates. About 50% of the Delmarva croplands either receive animal manure or sewage sludge or are planted to soybean annually (Staver and Brinsfield, 1995a).

Another factor influencing the effectiveness of strategies designed to reduce sediment and sediment-bound P in surface water runoff is the long-term buildup of soil P levels (National Research Council, 1993; Carpenter et al., 1998). Practices such as reduced tillage and N-based NMPs, particularly for animal manure and sewage sludge application, result in P application rates several times greater than crop P-removal rates. Long-term studies comparing total P losses in continuous corn production systems indicate elevated dissolved P losses in surface water runoff from fields that are not tilled (Staver and Brinsfield, 1995b). The potential for dissolved P transport is strongly correlated with soil P levels, which are a function of the differences between application rates and crop removal rates (Sharpley, 1995). Most soils in Maryland, Delaware, and Pennsylvania have P levels greater than that needed for optimal crop production, and recent trends show increasing soil test P levels, particularly on fields using N-based nutrient management plans for the application of animal manure and sewage sludge (Coale, 1999; Sims, 1999).

The effectiveness of erosion-based strategies began to be questioned in the early 1990s (National Research Council, 1993) when it was recognized that the soil loss–nutrient transport relationships used in projecting nutrient loads from agricultural land probably overestimated reductions that would be achieved using the proposed nonpoint-source control strategy. The 1991 and 1997 reevaluations found that nonpoint loadings of N to the Bay had yet been little reduced (Chesapeake Bay Program, 1991, 1997), but it was assumed that soil conservation practices were more effective in reducing nonpoint P loadings. However, recent evidence of a gradual increase in P levels in agricultural soils places in some doubt the degree to which significant reduction of P losses from croplands in the watershed have indeed been achieved. The disparity in the 1997 reevaluation (Chesapeake Bay Program, 1997) between model predictions of decreased loadings from Coastal Plain agricultural watersheds and unchanged or increasing N and P concentrations observed in the tidal subestuaries into which they drain raises similar questions about the efficacy of BMPs. For example, BMPs for P still depend on no-till practices even though, as discussed below, research indicates that continuous no-till can actually increase total P losses.

This disparity between projected and achieved changes in nonpoint-source nutrient loads is underscored by findings from the German Branch watershed, located in the upper Choptank River drainage basin. This watershed is predominately agricultural and was targeted for aggressive implementation of BMPs in 1990. The project has generated tremendous interest among farmers in the watershed, resulting in high levels of implementation of NMPs and SWCPs and considerable cost-sharing for installation of BMPs. Despite high levels of implementation, N concentrations increased in German Branch from 1990 to 1995 and P concentrations have remained at elevated levels, but changing of cropping practices further complicated assessment of BMP effectiveness (Millard et al., 1997).

Pfiesteria, Animal Wastes, and Mandated Nutrient Management

During the summer of 1997, fish with lesions were found and fish kills observed in the Pocomoke River and tributaries of the Manokin River and Fishing Bay (Fig. 1), which drain Coastal Plain watersheds with intense agriculture. The toxin-producing dinoflagellate *Pfiesteria piscicida* (discussed further under Emerging Issues) and similar species were implicated as the probable causative factor. Experiments and observations in North Carolina, where toxic *Pfiesteria* outbreaks have been more extensive, suggested that such outbreaks are made more likely or severe by enrichment with organic matter or nutrients (Burkholder and Glasgow, 1997). Therefore, much attention was focused on nutrient loadings, particularly from the intensive grain and poultry production on the Delmarva Peninsula (Hughes, 1997). Concern over the toxic *Pfiesteria* outbreaks was further heightened by the documentation of short-term memory impairment and other health effects in humans apparently exposed to toxic *Pfiesteria* outbreaks (Grattan et al., 1998). This has cast an intense spotlight on the overall effectiveness of agricultural programs aimed at reducing nutrient loads to Chesapeake Bay, particularly with regard to land application of poultry manure.

Nutrient inputs into the Bay along its eastern shore are dominated by agricultural sources. For example, the Chesapeake watershed model estimates that 60% of the N inputs from the Maryland eastern shore to the Bay are from agriculture, about half of that amount from animal manure. For the Pocomoke River watershed, approximately 70% of the P and 74% of the N loads are from agricultural sources (Hughes, 1997). The watershed has one of the highest concentrations of poultry production in the region. Surveys indicate that the large quantity of manure generated in the watershed is applied to 42% of the cropland annually. Because poultry manure is relatively rich in P relative to the N to P requirements of crops, repeated manure application may result in elevated P concentrations in soils and, therefore, increased dissolved P in surface runoff (Sharpley, 1995; Staver and Brinsfield, 1995b; Carpenter et al., 1998).

Although a direct link has not been established between the growth and toxicity of *Pfiesteria*, a heterotroph, and nutrient inputs, toxic outbreaks (Burkholder and Glasgow, 1997) and higher densities of *Pfiesteria*-like dinoflagellate zoospores (Pinckney et al., 2000) in North Carolina have been found mainly under hypereutrophic conditions. Based on the limited studies available at the time, evidence regarding nutrients and other environmental conditions where the toxic outbreaks occurred, and concerns about the effectiveness of the N-based BMPs that were being applied, groups of scientific experts advised a commission established by Maryland's Governor that reducing nutrient loads would probably lower the risk of future outbreaks of toxic algal blooms and suggested approaches for more effective nutrient management in agriculture (Hughes, 1997). The recommendations of this commission led to the passage of the nation's first mandatory nutrient management law, Maryland's Water Quality Improvement Act of 1998, which requires the development and implementation of N- and P-based nutrient management plans for most farms in the state by 2005. Virginia and Delaware followed suit in 1999 by passage of mandatory nutrient management laws also requiring P-based nutrient management where animal manure is applied to fields.

Toward More Effective Controls

Although the connection between poultry manure and *Pfiesteria* outbreaks remains subject to scientific uncertainty, the lack of overall progress in documented water quality improvements has raised serious questions regarding the effectiveness of current agricultural strategies in achieving the reductions in both N and P loadings needed to achieve existing water quality objectives. Clearly, the measure of progress is shifting from *process* assessment, such as number of BMPs installed or area of farmland with nutrient management plans, to *outcome* assessment of actual reductions in nutrient losses that result in demonstrable changes in water quality. Absent such indicators showing progress, pressure for additional regulatory controls on agriculture will continue to increase. Some opportunities for improving success in agricultural nutrient source reductions are discussed below for N and P.

For soil types not predisposed to high rates of erosion, such as those on the lower Delmarva Peninsula, reductions in surface runoff of P will require better management of P levels in the uppermost soil horizons. However, one of the most effective and widely promoted BMPs, no-till, actually increases P levels in the top soil horizons and under certain conditions results in increased dissolved P losses compared with conventionally tilled fields receiving the same P-fertilizer rates (Staver and Brinsfield, 1995b). Therefore, for those fields with extremely high near-surface soil P levels and low erosion potential, periodic tillage could have the immediate effect of significantly reducing dissolved P losses.

In the long term, however, better management of P inputs on a watershed basis will be required. For soils where P levels are above those required for plant growth, P fertilization rates could be reduced without affecting crop yields. Although this approach is straightforward for fields using inorganic fertilizers, it becomes more problematic for cropland receiving animal manure or sewage sludge. For example, the ratio of N to P in the crops is approximately 6:1 on a weight basis or more than double the plant-available ratio in poultry manure (Sims, 1987) and most sewage sludge applied to cropland (Staver and Brinsfield, 1995b). Repeated applications of poultry manure in the context of nutrient management plans based on crop-N requirements result in P application rates several times greater than annual crop removal rates and consequently a buildup of soil P levels.

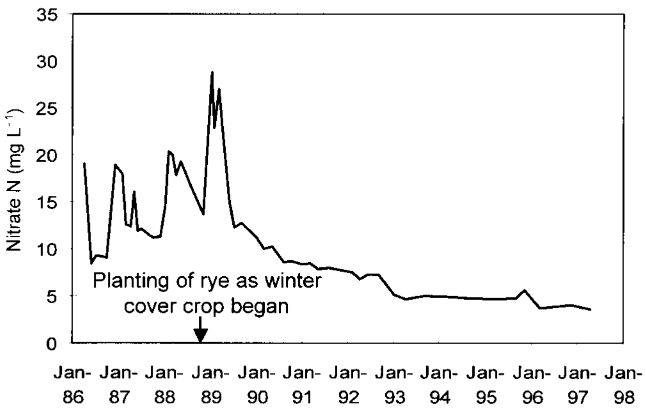

Fig. 6. Ground water nitrate levels under an experimental cornfield in Maryland's Coastal Plain managed for realistic crop yields with split nitrogen applications, no-till, and grassed waterways, showing the long-term effect of winter cover crop plantings (Staver and Brinsfield, 1998).

The P content of poultry manure may be reduced through nutritional technology and manure management. One approach includes mixing the enzyme phytase in feed to convert unavailable organic P into a form that poultry can use, thereby reducing the amount of inorganic P added to feed. Use of phytase could reduce manure P output by 25% (Hughes, 1997) resulting in N to P ratios more closely matching crop removal rates.

To the extent that these strategies do not balance P nutritional requirements and water quality goals on a watershed basis, other approaches to utilize the P in animal manure and sewage sludge must be considered, including composting, direct combustion, and biomass production for co-generating electricity (Hughes, 1997). Ultimately, the goal should be to stabilize P budgets by substituting organic P for inorganic P, thereby reducing the importation of P from outside the watershed.

Until recently, the long-term strategy to reduce N losses from agriculture has similarly focused mainly on limiting surface water runoff and matching N inputs to crop needs. However, unlike P, erosion-controlling BMPs provide only marginal opportunities for reducing N losses since the major flowpath for N losses is leaching to ground water during winter recharge (Staver and Brinsfield, 1998). Ground water nitrate N concentrations >10 mg L^{-1} have been routinely reported in cash-grain producing regions within the Bay watershed (Bachman, 1984; Weil et al., 1990). Furthermore, stream water quality monitoring (Bachman and Phillips, 1996) and ground water seepage patterns (Reay et al., 1992; Staver and Brinsfield, 1996) indicate that large quantities of nitrate N entering ground water are eventually discharged into surface waters.

As with P, the problem is exacerbated by the application of animal manure, sewage sludge, and the increasing area planted to soybean, all of which result in elevated nitrate concentrations in ground water even with nutrient management planning (Angle, 1990; Staver and Brinsfield, 1995a). These limitations, coupled with potential droughts that limit plant N uptake and mineralization from the organic pool (Staver and Brinsfield, 1990), suggest that other strategies will be necessary to meet N-reduction goals from agricultural sources.

Ground water recharge and nitrate leaching for most regions occur mainly in the fall and winter months when crop uptake and evaporation is at its minimum. Therefore, the most important factor that determines nitrate leaching is the pool of nitrate available in the root zone just prior to the onset of ground water recharge (Staver and Brinsfield, 1998).

Historically, winter cereal cover crops have been used as a cropping practice primarily to reduce soil erosion and improve soil physical properties. However, more recently studies have demonstrated the value of cover crops in minimizing nitrate losses to ground water and improving the sustainability of modern agriculture (Meisinger et al., 1991). From a water quality perspective, the general underlying principle is that winter cover crops can directly affect ground water quality by reducing the pool of soil nitrate available for leaching at the beginning of the ground water recharge cycle (Staver and Brinsfield, 1995a). In Maryland's Coastal Plain, cereal rye (*Secale cereale* L.), planted as a cover crop following corn harvest, consistently reduced nitrate in root zone leachate to <1 mg L^{-1} during most of the ground water recharge period (Staver and Brinsfield, 1998). This reduced annual nitrate leaching losses by approximately 80% relative to winter-fallow conditions. Shallow ground water nitrate N concentrations under long-term continuous corn production decreased from 10 to 20 mg L^{-1} to <5 mg L^{-1} after seven years of cover crop use (Fig. 6). Furthermore, use of cover crops following soybean harvest and the application of animal manure and sewage sludge significantly reduced nitrate leaching losses relative to winter fallow treatments (Staver and Brinsfield, 1995a).

As a result of their ability to reduce nitrate leaching, cereal winter cover crops are increasingly becoming a major component of strategies for nutrient reduction in the Chesapeake Bay watershed, particularly around tidal tributaries where the dominant land use is agriculture. Unfortunately, farmers' willingness to integrate cover crops into their farm management plan has been limited due to their short-term added cost. However, stimulated both by concerns about the effects of nutrients on *Pfiesteria* outbreaks and the large amount of residual N in topsoil resulting from summer drought conditions during 1997, the state of Maryland increased its financial support for cover crop implementation.

EMERGING ISSUES

As the restoration of the eutrophic Chesapeake Bay proceeds, a number of issues in addition to those related to agricultural nonpoint-source controls have emerged that pose interesting challenges and opportunities for both science and management:

- Linking eutrophication and living resources. Restoration of living resources is a publicly supported goal for the Chesapeake Bay restoration, but, as discussed above, the effects of eutrophication on living resources are poorly quantified. The Chesapeake 2000 Agreement calls for the determination of the conditions needed to protect aquatic living resources and on that basis to develop water quality standards (probably for dissolved oxygen, chlorophyll, and water clarity) that will serve as the basis for nutrient load allocation for various parts to the Bay. The objective is to meet these standards by 2010. However, the relationship of eutrophication to living resources is far more complicated than reflected in oxygen conditions, phytoplankton standing stock, and water clarity. Exactly how will the living resources be affected by reversal of eutrophication and what are the implications for multispecies management in the context of this ecosystem restoration?

- Reducing atmospheric deposition. Atmospheric deposition is an important but incompletely understood source of nutrients for the Chesapeake Bay, particularly N, for which atmospheric deposition accounts for about one-fourth of total loadings (Fisher and Oppenheimer, 1991; Valigura et al., 1996). Regionally, atmospheric deposition is the most important contributor of nitrogen runoff in the northeast USA (Jaworski et al., 1997). The fixed N deposited from the atmosphere on the Bay and its watershed include oxides of N resulting from combustion of fossil fuels locally (for example, from automobiles) and remotely (for example, from coal-burning power plants in the Ohio Valley, well outside of the Bay's watershed) and ammonia volatilized from animal wastes. The increasing releases of fixed N into the atmosphere are an important manifestation of the substantial human alteration of the global N cycle (Vitousek et al., 1997). This alteration contributes not only to coastal eutrophication but also to degraded air quality (through formation of photochemical smog), global warming (through production of nitrous oxide), loss of soil fertility and terrestrial plant biodiversity, and acidification of streams and lakes. Environmental scientists are presented the challenge of understanding the interrelationships among these various effects, while environmental managers are presented the opportunity to find common solutions to problems of air quality (ground-level ozone) and coastal eutrophication.
- Enhancing nutrient sinks. It is becoming increasingly clear that nutrients emanating from nonpoint sources, be they from agriculture, atmospheric deposition, or urban runoff, cannot be adequately controlled at their sources. Nutrient reduction must also be addressed by increasing the capacity and effectiveness of nutrient sinks that trap both dissolved and particulate nutrients, incorporate nutrients into long residence-time biomass, or—in the case of nitrogen—result in their conversion to inert gas. It should be feasible, through both more effective nutrient management on the field and strategic restoration of forested and grass buffers and wetlands, to reduce agricultural nutrient inputs to the estuary by 50% without severe economic effects (Boesch and Brinsfield, 2000). Although protection of wetlands and restoration of riparian forests have long been part of environmental management, the concept of landscape management to optimize nutrient sinks presents new challenges and opportunities for both science and management (National Research Council, 1993).
- Controlling sprawling suburban development. The agreement for 40% nutrient reduction also involves the commitment to maintain the targeted nutrient loads as caps, in perpetuity. Yet, the human population continues to grow in the Chesapeake watershed and, perhaps more importantly, the conversion of forested and agricultural lands to development has grown at a rate two to three times greater than population growth. More sewage is produced and nonpoint sources of nutrients generally increase as a result of land development, power generation, tailpipe emissions, and horticultural fertilization. In order to sustain accomplishments in reversing eutrophication, more effective treatment of wastes and limitations on land development will be required. Growth management, or *smart growth*, to give it a more positive sounding name, is being addressed in a number of the political jurisdictions in the Chesapeake Bay watershed and is addressed in the land use goals of the 2000 Agreement. This presents opportunities for productive symbiosis between landscape ecologists and urban planners to optimize environmental, social, and economic benefits.
- Predicting and preventing harmful algal blooms. The Chesapeake has been fortunate in that increased phytoplankton production resulting from eutrophication has stimulated few blooms of toxic or noxious algae that increasingly plague other regions. This perception changed with the recent implication of *Pfiesteria piscicida* or similar toxin-producing dinoflagellates as the probable causative factor of fish kills and lesions in limited regions of the Bay. The so-called phantom dinoflagellate, *Pfiesteria*, has a complex life history in which the organism can metamorphose into as many as 24 forms (including toxic and nontoxic zoospores, amoebae, and cysts) depending on, among other factors, exposure to fish (Burkholder and Glasgow, 1997). Toxin-producing stages release potent toxins, which erode the skin of fish, stun and, at high enough concentrations, kill them. The heterotrophic dinoflagellates then feed on fish tissue. Particularly because of the risks to human health (Grattan et al., 1998), *Pfiesteria* in the Chesapeake Bay was catapulted into international headlines. The great public attention has provided the opportunity to increase understanding of the qualitative effects of nutrients on plankton communities, including other noxious and nuisance blooms of algae, and, as discussed earlier, to address agricultural nonpoint source controls more aggressively.

CONCLUSION

The essential core of the restoration of the Chesapeake Bay ecosystem is the reversal of eutrophication in order to reduce hypoxia, increase water clarity, improve habitat quality, and foster trophic structures conducive to the production of valued living resources. The public and political commitment to undertake this task was informed by scientific hypotheses, which have since been enriched and confirmed. Nonetheless, there remain many challenges for science to refine strategic models, develop effective source controls, and decipher the responses of this complex ecosystem to what is, in effect, a large experiment. Progress toward nutrient reduction goals has been made; however, reducing nonpoint sources of nutrients mobilized by human activities, particularly from agriculture and fossil fuel combustion (which together account for well over half of the nutrients reaching the Bay), remains the most elusive challenge for management. This will require both reductions of applications or emissions of nutrients and more intelligent landscape management to optimize nutrient reservoirs and sinks.

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Hypoxia in the Gulf of Mexico

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ABSTRACT

Seasonally severe and persistent hypoxia, or low dissolved oxygen concentration, occurs on the inner- to mid-Louisiana continental shelf to the west of the Mississippi River and Atchafalaya River deltas. The estimated areal extent of bottom dissolved oxygen concentration less than 2 mg L^{-1} during mid-summer surveys of 1993–2000 reached as high as 16 000 to 20 000 km^2. The distribution for a similar mapping grid for 1985 to 1992 averaged 8000 to 9000 km^2. Hypoxia occurs below the pycnocline from as early as late February through early October, but is most widespread, persistent, and severe in June, July, and August. Spatial and temporal variability in the distribution of hypoxia exists and is, at least partially, related to the amplitude and phasing of the Mississippi and Atchafalaya discharges and their nutrient flux. Mississippi River nutrient concentrations and loadings to the adjacent continental shelf have changed dramatically this century, with an acceleration of these changes since the 1950s to 1960s. An analysis of diatoms, foraminiferans, and carbon accumulation in the sedimentary record provides evidence of increased eutrophication and hypoxia in the Mississippi River delta bight coincident with changes in nitrogen loading.

THE inner- to mid-continental shelf (depths of 5 to 60 m) of the northern Gulf of Mexico, from the Mississippi River birdfoot delta westward to the upper Texas coast, is the site of the largest zone of hypoxic bottom water in the western Atlantic Ocean coastal zone (values in Rabalais et al., 1999; Rabalais and Turner, 2001; cf. Boesch and Rabalais, 1991). Hypoxia is operationally defined as dissolved oxygen levels less than 2 mg L^{-1}, or ppm, for the northern Gulf of Mexico, because that is the level below which trawlers usually do not capture any shrimp or demersal fish in their nets (Pavela et al., 1983; Leming and Stuntz, 1984; Renaud, 1986). The areal extent of this zone, with estimates up to 20 000 km^2 of near-bottom waters with dissolved oxygen levels $<$2 mg L^{-1}, rivals the largest hypoxic areas elsewhere in the world's coastal waters, namely the Baltic Sea and the northwestern shelf of the Black Sea. The northern Gulf of Mexico is strongly influenced by the Mississippi and Atchafalaya Rivers, whose combined discharges account for 80% of the total freshwater input (calculated from U.S. Geological Survey streamflow data for 37 U.S. streams discharging into the Gulf of Mexico; Dunn, 1996). 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 and nutrient fluxes (Pokryfki and Randall, 1987; Justić et al., 1993; Rabalais et al., 1996; Wiseman et al., 1997). The freshwater fluxes dictate, along with climate, a strong seasonal pycnocline. Nutrients delivered by the rivers support high primary production (Sklar and Turner, 1981; Lohrenz et al., 1990, 1994, 1997), of which approximately 50% fluxes to bottom

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