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Challenges and Opportunities for Science in Reducing Nutrient Over-enrichment of Coastal Ecosystems

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ABSTRACT: Nutrient over-enrichment has resulted in major changes in the coastal ecosystems of developed nations in Europe, North America, Asia, and Oceania, mostly taking place over the narrow period of 1960 to 1980. Many estuaries and embayments are affected, but the effects of this eutrophication have been also felt over large areas of semi-enclosed seas including the Baltic, North, Adriatic, and Black Seas in Europe, the Gulf of Mexico, and the Seto Inland Sea in Japan. Primary production increased, water clarity decreased, food chains were altered, oxygen depletion of bottom waters developed or expanded, seagrass beds were lost, and harmful algal blooms occurred with increased frequency. This period of dramatic alteration of coastal ecosystems, mostly for the worse from a human perspective, coincided with the more than doubling of additions of fixed nitrogen to the biosphere from human activities, driven particularly by a more than 5-fold increase in use of manufactured fertilizers during that 20-year period. Nutrient over-enrichment often interacted synergistically with other human activities, such as overfishing, habitat destruction, and other forms of chemical pollution, in contributing to the widespread degradation of coastal ecosystems that was observed during the last half of the 20th century. Science was effective in documenting the consequences and root causes of nutrient over-enrichment and has provided the basis for extensive efforts to abate it, ranging from national statutes and regulations to multi-jurisdictional compacts under the Helsinki Commission for the Baltic Sea, the Oslo-Paris Commission for the North Sea, and the Chesapeake Bay Program, for example. These efforts have usually been based on a relatively arbitrary goal of reducing nutrient inputs by a certain percentage, without much understanding of how and when this would affect the coastal ecosystem. While some of these efforts have succeeded in achieving reductions of inputs of phosphorus and nitrogen, principally through treatment of point-source discharges, relatively little progress has been made in reducing diffuse sources of nitrogen. Second-generation management goals tend to be based on desired outcomes for the coastal ecosystem and determination of the load reductions needed to attain them, for example the Total Daily Maximum Load approach in the U.S. and the Water Framework Directive in the European Union. Science and technology are now challenged not just to diagnose the degree of eutrophication and its causes, but to contribute to its prognosis and treatment by determining the relative susceptibility of coastal ecosystems to nutrient over-enrichment, defining desirable and achievable outcomes for rehabilitation efforts, reducing nutrient sources, enhancing nutrient sinks, strategically targeting these efforts within watersheds, and predicting and observing responses in an adaptive management framework.

Introduction

The infusion of nutrients (particularly nitrogen [N] and phosphorus [P]) from the land via runoff or the deep ocean via upwelling is the reason that coastal ecosystems are highly productive and support most of the world's fisheries. Over a short period of time during the 1970s and 1980s it was discovered that large increases in loadings of nutrients from land as a result of human activities were changing many coastal ecosystems around the world, often for the worse. The rapid growth of scientific investigation and publication on coastal over-enrichment (Nixon 1995) was not only a result of awakening and expansion of research, but also of the concomitant rapid increase in the supply of organic matter in the water bodies stimulated by the increased nutrient loading (eutrophication by Nixon's definition) that was taking place during the last half of the 20th century.

Science has done a commendable job of documenting coastal eutrophication, identifying its root causes, and pointing out its actual and potential consequences (Rabalais 2002). As a result, major social and political commitments have been marshaled in many parts of the world to reduce nutrient over-enrichment. The increased supply of nutrients from land is now recognized as one of the most pervasive causes of the degradation of coastal ecosystems (National Research Council [NRC] 2000).

Although it was not necessarily easy for the scientists engaged in this transition, the path and recognition of the problem through commitment to alleviate it happened quite quickly in the course of human events. When I moved from the Chesapeake Bay region in 1980, debates were unresolved about the effects of nutrient over-enrichment on

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the bay as a whole and whether N was at all a factor. When I returned to the region in 1990, I found a large intergovernmental establishment pursuing a nearly single-minded goal of reducing both N and P inputs by 40% by the year 2000. While working on the Gulf of Mexico coast I helped initiate a research program with Nancy Rabalais and Gene Turner in 1985 on the scope and causes of hypoxia on the Louisiana inner continental shelf. To say that our hypothesis that hypoxia was linked to increased loadings of nutrients from the Mississippi River was greeted with skepticism by agencies and regional scientists alike is putting it mildly. It is just a natural phenomenon we were told and, besides, it would never be politically possible to reduce nutrient inputs in the heartland of American agricultural production, so far away from the Gulf. Fifteen years later, eight federal agencies, nine states, and two tribal governments agreed to a plan to reduce the extent of hypoxia to a level that would probably require reduction of upstream N inputs by 30% (Rabalais et al. 2002).

So what is next for science? The hard part, really. Scientists had observations, reconstructions from the sediment record, and general theories grounded on experiences elsewhere to develop a convincing picture of the course of coastal eutrophication. As we move into the less traveled territory of defining achievable states in the rehabilitation of over-enriched ecosystems, forecasting the trajectory of rehabilitation, developing the most effective means to reduce nutrient inputs, and observing progress in a highly variable world, scientists will have to be even more resourceful and holistic in their approaches.

This paper addresses challenges and opportunities for science as society moves forward to reducing nutrient over-enrichment of coastal ecosystems. I review the nature, scope, and history of coastal eutrophication, programmatic efforts underway to manage the alleviation of nutrient over-enrichment, and approaches for its abatement. This review sets the stage for the challenges that remain.

Eutrophication of Coastal Ecosystems

Manifestations

In his review of evolving concepts of coastal eutrophication, Cloern (2001, p. 226) used the term eutrophication in the sense of "the myriad biogeochemical and ecological responses, either direct or indirect, to anthropogenic fertilization of ecosystems at the land-sea interface." These responses can be direct or indirect, acute or chronic, subtle or profound, and beneficial or detrimental to human interests depending on the circumstances.

Subject to light limitation, the greater availability of N and P increases primary production of phytoplankton and, in some cases, benthic macroalgae or macrophytes. The resulting increase in phytoplankton biomass commonly reduces water clarity and results in greater delivery of organic matter to bottom waters and the seabed. Nutrient enrichment also results in qualitative changes in the dominant primary producers, both in the water column and photic zone benthic environments. This may cause intense blooms, including harmful algal blooms that produce toxins or nuisance conditions. Increased organic production and nutrient supply also stimulate microbial populations and processes, making the microbial loop a more prominent trophic pathway.

The enhanced organic production may result in increased secondary production that extends through food chains to exploited fishery populations, particularly of pelagic species (Caddy 1993, 2000), but this may not always be the case (Micheli 1998). On the other hand, decomposition of the greater supplies of organic matter often results in depletion of dissolved oxygen (hypoxia) in stratified bottom waters to levels too low to sustain fishes and invertebrates. Diminished water clarity and stimulated epiphytic growth may eliminate seagrass and microalgal meadows or coral reefs that provide important habitats. Eutrophication that proceeds to the point of producing seasonally persistent hypoxia and extensive loss of shallow vegetated habitats produces obvious negative effects on fishery resources.

Distribution and Susceptibility

These various manifestations of eutrophication are widely evident in coastal ecosystems of developed nations, including North America, Europe, Asia, and Australia. They are less well known in the developing world because of lack of study, lower loadings of nutrients at their present stage of development, or different sensitivities of tropical coastal ecosystems (Corredor et al. 1999). Semi-enclosed seas and large estuaries are affected by hypoxia, seagrass losses, and algal blooms over surprisingly large areas (Fig. 1). This includes deep-basin hypoxia and algal blooms extending over the entire southern Baltic Sea (Jansson and Dalhberg 1999; Elmgren 2001; Elmgren and Larsson 2001), the virtual elimination of the 10,000 km^2 meadow of the red alga *Phyllophora* on the northwestern shelf of the Black Sea (Zaitsev 1999), algal blooms and anoxia in the northern Adriatic Sea (Malone et al. 1999), and a 20,000 km^2 area of seasonal bottom-water hypoxia off the Mississippi and Atchafalaya rivers in the northern Gulf of Mexico (Com-

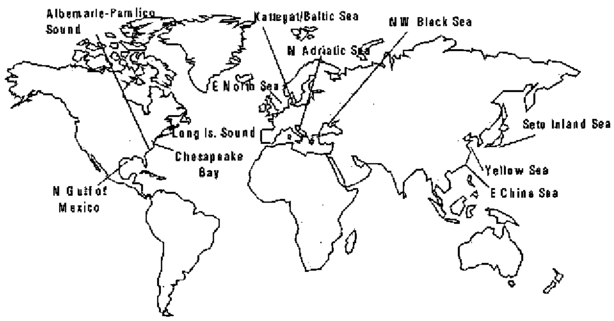

Fig. 1. Locations of regions of large-scale nutrient over-enrichment.

mittee on Environment and Natural Resources 2000; Rabalais et al. 2002).

Many smaller bays, estuaries and lagoons in the United States (Bricker et al. 1999), Europe (Crouzet et al. 1999; Conley et al. 2000), Japan (Suzuki 2001), and Australia (McComb 1995) have been significantly modified by nutrient over-enrichment. For example, 44 of 138 bays and estuaries along the coast of the U.S. were assessed as having high levels of eutrophication and an additional 40 have moderate symptoms (Bricker et al. 1999). Fully 67% of the combined surface area of these bays and estuaries exhibit moderate to high degrees of increased phytoplankton growth, increased growth of macroalgae and epiphytes, low dissolved oxygen, harmful algal blooms, and loss of seagrass.

It is important to keep in mind that not all coastal ecosystems are equally susceptible to nutrient over-enrichment. Cloern (2001) contrasts Chesapeake Bay, which is particularly sensitive to eutrophication, with San Francisco Bay, which is not. San Francisco Bay receives higher N and P loading rates than Chesapeake Bay, but has lower standing stocks of phytoplankton and little or no hypoxia as a result of its vigorous tidal mixing and light limitation caused by sediment resuspension. The relative susceptibility of coastal ecosystems depends on dilution, water residence time and flushing, stratification, suspended material load or light extinction, among other factors (NRC 2000). This susceptibility can be classified according to the dilution and flushing by freshwater inflow and tidal prism volumes (Dettmann 2001) to yield indicators that are useful for protection from nutrient over-enrichment.

Much attention has been focused on the anthropogenic inputs of N as the principal cause of eutrophication in coastal ecosystems in contrast to freshwater ecosystems, which tend to be P-limited. The processes that lead to the preponderance of N limitation in estuarine and coastal ecosystems are reviewed by NRC (2000). In efforts to reduce nutrient over-enrichment it is important to keep in mind that loadings of both N and P are many-times elevated over background levels for many coastal ecosystems; nutrient limitation varies based on the different biogeochemistry within and between ecosystems and, often, seasonally; and reducing the inputs of one nutrient has consequences for the fate of the other.

Phytoplankton production tends to be P-limited in tidal freshwater reaches of estuaries. Reducing P loading to these subsystems can result in increasing the export of N, not taken up by phytoplankton, to the lower estuarine reaches. Reducing N loading, without reducing P loading, may stimulate N fixation by harmful or noxious cyanobacteria. Nutrient limitation can also shift seasonally with P limitation in spring and N limitation in summer (Fisher et al. 1999; Conley 2000). Shallow tropical ecosystems characterized by carbonate sediments tend to be P rather than N limited (Corredor et al. 1999). Finally, the type of phytoplankton production stimulated is influenced by the ratios of available nutrients (Justić et al. 1995), not only of N and P, but also silicon (which human activities tend to decrease) and, potentially, other micronutrients or the chemical form of macronutrients (e.g., organic N). For all these reasons, it is important that efforts to reduce the deleterious effects of nutrient over-enrichment integrate N and P controls rather than just focus on N.

Explosive Increase in Late 20th Century

Although human activities have altered the delivery of nutrients to coastal waters as long as the human species has been around, these fluxes have been successively increased as a result of land clearing for hunting and agriculture, population growth, industrial development, increasing combustion of fossil fuels, human interest in cleanliness, and the green revolution in agricultural production fueled by mined and manufactured fertilizers. The eutrophication of some coastal ecosystems had already advanced by the beginning of the 20th century (Conley 2000). The effects of increased delivery of nutrients and sediments that resulted from agricultural land clearing by European colonists are clearly evident in the increased accumulation of organic carbon and biogenic silica in Chesapeake Bay sediments (Cooper 1995). Increased deposition of biogenic silica on the Louisiana continental shelf occurred during the 19th century, also coincident with conversion of land within the Mississippi River basin to agriculture (Rabalais et al. 1996).

During the early 20th century population growth and industrialization led to intense loading of organic wastes in many coastal ecosystems. The

TABLE 1. Late 20th century manifestations of nutrient over-enrichment in major coastal ecosystems in Europe, North America, Japan, and Australia. Except where noted information from Cloern (2001).

Coastal Ecosystem	Manifestation	Period
Baltic Sea area	Nitrogen supply to Kattegat doubled (Andersson and Rydberg 1988)	1950–1980
	Nitrogen supply to Lanholm Bay tripled (Rosenberg et al. 1990)	1960–1980
	Transparency in Åland archipelago decreased by 50%	1984–1994
	Primary production in the Kattegat doubled	1960–1980s
	Area of "dead bottom" in deep Baltic tripled (Jansson and Dahlberg 1999)	1950s–1980s
	Bottom dissolved oxygen levels in Kattegat decreased by one-fourth	1960s–1980s
North Sea and Wadden Sea	Nanoplankton biomass at Helgoland tripled (Colijn and Reise 2001)	1970s–1980s
	Phytoplankton biomass and primary production in Wadden Sea doubled	1970–1980
	Duration of nondiatom blooms in Wadden Sea increased by factor of more than four	1975–1985
	Macrobenthos mean weight of individual in Wadden Sea decreased by 50%	1970–1980
	Macrobenthos biomass in Wadden Sea doubled	1980–1990
Northern Adriatic Sea	PO_4 and dissolved inorganic nitrogen concentrations in Po River more than doubled (Harding et al. 1999)	1970s–1980s
	Primary production doubled	1960–1970
	Average summer oxygen concentration in bottom layer declined by 37% (Justić et al. 1987)	1943–1984
	Hypoxia first developed (Justić et al. 1987)	1975
Northwestern Black Sea	Inorganic phosphate discharges from Danube River increased 50%, inorganic nitrogen discharges increased by factor of 4 (Mee 2001)	1960–1980s
	Transparency decreased by >50% (Zaitsev 1999)	1960s–1970s
	Phytoplankton biomass increased 40-fold along Romanian coast (Mee 2001)	1963–1983
	Phyllophora algal meadow decreased by 70% (Zaitsev 1999)	1950s–1980s
	Large-scale hypoxia first observed (Zaitsev 1999)	1973
Chesapeake Bay	NO_3 in Potomac River inflow increased 2.5 times (Zimmerman and Canuel 2000)	1960s–1980s
	Phytoplankton biomass tripled	1960s–1970s
	Centric:pennate diatom ratio in sediment record doubled (Cooper 1995)	1950s–1980s
	Organic carbon incorporation into sediments increased by 50% (Cornwell et al. 1996)	1960s–1980s
	Organic enrichment sediment biomarkers increased by factor of three (Zimmerman and Canuel 2000)	1970s–1980s
	Volume of hypoxic bottom water doubled (Hagy 2002)	1970s–1980s
	Hypoxia-tolerant benthic Foraminifera doubled in dominance (Karlsen et al. 2000)	1960s–1980s
	Seagrass extent reduced by 80% (Orth and Moore 1983)	1970–1980
Northern Gulf of Mexico	Flux of nitrate from Mississippi River Basin tripled	1960s–1980s
	Incorporation of organic carbon in sediments doubled (Rabalais et al. 1996)	1950s–1980s
	Incorporation of biogenic silica in sediments doubled (Rabalais et al. 1996)	1960s–1980s
	Hypoxia-tolerant benthic Foraminifera increased in dominance (Rabalais et al. 1996)	1950s–1980s
Tampa Bay	Phytoplankton biomass doubled (Johansson and Greening 2000)	1950s–1970s
	Seagrass extent reduced by 50% (Johansson and Greening 2000)	1950s–1980s
Mikawa Bay	Transparency decreased by more than one third (Suzuki 2001)	1960–1970
	Days red tides observed tripled (Sukuki 2001)	1970–1983
Seto Inland Sea	Number of red tides increased by factor of >4	1960–1975

enrichment effects of organic waste disposal were sometimes severe, but were relatively localized. Waste treatment to degrade organic matter helped to alleviate these localized effects, such as dissolved oxygen sags in estuaries (e.g., in the Thames and Delaware Rivers), but did little to reduce the inputs of N and P, which are mineralized in the process. At the same time, the use of phosphate-based detergents compounded the growing eutrophication, particularly in freshwater ecosystems, including the tidal freshwater reaches of estuaries such as the Potomac River (Jaworski 1990). It was not until after World War II that the loadings resulting from fossil fuel combustion and fertilizer use grew rapidly to overwhelm the other sources, causing increases in river exports of N of 2 to 20 times that of the prehistoric levels for coastal regions of eastern North American and Western Europe (Howarth et al. 1996).

The explosive and synchronous intensification of eutrophication in susceptible coastal ecosystems in North America, Europe, and Japan is striking (Table 1). Within two decades (roughly 1960 to 1980), large coastal ecosystems were substantially changed, as areas of hypoxia developed or greatly intensified, phytoplankton production or biomass doubled, and benthic macrophyte meadows contracted. The coincidence of this period with the rapid growth of world fertilizer consumption and emission of nitrogen oxides from fossil fuel combustion is compelling evidence of the primary driving forces of coastal eutrophication (Fig. 2). The total N fixed by human activities more than doubled between 1960 and 1980 and now probably ex-

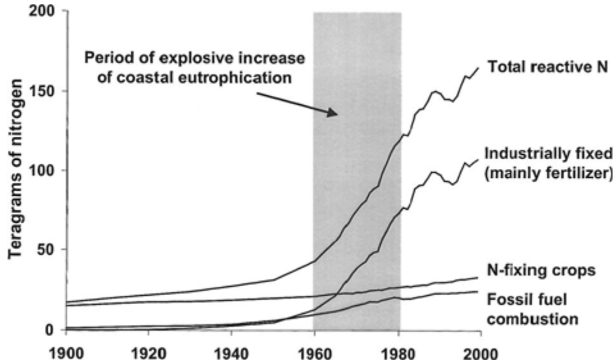

Fig. 2. Period of the explosive increase in coastal eutrophication in relation to global additions of anthropogenically fixed nitrogen (fixation estimates provided by Galloway, personal communication).

ceeds the total fixed by natural processes in the biosphere (Vitousek et al. 1997). The increase in N fertilizer use slowed after 1980 in North America, Europe, and Japan. Most of the global increase in fertilizer use after 1980 took place in developing nations (Tilman et al. 2001).

Although the nutrient inputs into urbanized coastal ecosystems such as Narragansett Bay (Nixon 1997) and Long Island Sound (Long Island Sound Study 1998) are dominated by municipal and industrial waste discharges and atmospheric deposition is a very important source of N in others, agriculture is the most important anthropogenic source of nutrients in most coastal ecosystems (Howarth et al. 1996). Controlling agricultural inputs of nutrients has been and will remain the most difficult challenge as food production grows to feed Earth's burgeoning population, projected to stabilize at more than 9 billion during this century (Nixon 1995; Tilman et al. 2001).

It should not be forgotten that humans have altered coastal ecosystems in many other ways in addition to nutrient over-enrichment. These include massive sedimentation from soil erosion; hydrological modification of both catchments and coastal environments; the direct (removal of consumers) and indirect (collateral habitat modification) effects of fishing; introductions of nonindigenous species; elimination or modification of habitats, such as wetlands, tidal flats, and oyster reefs, that serve not only as sites of food and refuge but also as natural nutrient sinks; and introduction of toxicants. Several or all of these maladies have degraded most coastal ecosystems (Elmgren 2001). Furthermore, many of these impacts interact, often synergistically.

The interaction of fishing (removing biological productivity) and eutrophication (stimulating it) deserves particular attention. Caddy (2000) describes instances where nutrient enrichment from human activities has increased fisheries production, particularly of pelagic planktivores. He also pointed out how removal of consumers through fishing may reduce grazing rates and exacerbate problems related to excess biomass of primary producers (e.g., reduced water clarity, hypoxia). Jackson et al. (2001) argued for the primacy of overfishing as the cause of the collapse of coastal ecosystems, in the sense that changes related to fishing preceded other impacts and preconditioned the susceptibility of ecosystems to these impacts. This seems to be clearly the case for coral reefs and kelp forests among other coastal ecosystems. Human-enhanced nutrient inputs also began more than a century ago. It is unlikely that the eutrophication-related collapses observed in some ecosystems (northwestern shelf of the Black Sea and Gulf of Mexico, for example) were necessarily preconditioned by removal of consumers controlling excess production from the top down.

Jackson et al. (2001) cited over-harvesting of filter feeding oysters in the Chesapeake Bay as setting the stage for subsequent changes in that ecosystem. It is implausible that restoration of the oyster populations alone could completely eliminate the adverse manifestations of nutrient over-enrichment without nutrient source reduction. Reducing bottom-up nutrient stimulation and increasing top-down controls through oyster restoration are, in fact, both objectives of the Chesapeake restoration strategy (Boesch et al. 2001b). This exemplifies the fact that with regard to the management and rehabilitation of coastal ecosystems, the concept of primacy is not very useful. Integrated goals and strategies are required.

Managing Coastal Eutrophication

Institutional Arrangements

Recognition of the impacts of nutrient over-enrichment has led to a variety of commitments and organized efforts to reverse eutrophication and rehabilitate the environmental quality and resources of degraded coastal ecosystems. These have run the gamut from national statutes and regulations to reduce emissions; multinational directives implemented at the national level; multijurisdictional compacts involving several nations or states, provinces, or prefectures; coastal management programs; and advisory and educational efforts.

During the 1970s mounting concerns about environmental degradation in the semi-enclosed seas surrounding Europe led to multinational conventions that have eventually incorporated commitments for pollution reduction. These are referred to as the Barcelona Convention (1975) for the

Mediterranean Sea, the Oslo (1972) and Paris (1974) Conventions for the North Sea, the Helsinki Convention (1974) for the Baltic Sea, and the Bucharest Convention (1994) for the Black Sea (Haas 1990; Crouzet et al. 1999; Mee 2001). Scientific and technical assessments are included among the activities pursuant to each of these conventions.

The Barcelona Convention, or Mediterranean Action Plan, includes a general commitment to take all appropriate measures to prevent, abate, and combat pollution of the Mediterranean Sea area, but has not advanced to include many specific goals for abatement of nutrient pollution (Crouzet et al. 1999). The Oslo Convention, which deals with discharges from ships and aircraft, and the Paris Convention, which addresses land-based sources of pollution, were combined in 1992 under the Oslo and Paris Commission (OSPAR). In 1987 ministers of the OSPAR nations (except for Great Britain) set the objective of reducing inputs of N and P by 50% by 1995 into areas where these inputs are likely, directly or indirectly, to cause pollution. In 1988 the parallel Helsinki Commission (HELCOM) adopted a very similar objective of halving the anthropogenic discharges of N and P over a 10-year period.

The European enclosed coastal sea conventions include the littoral states as signatories, e.g., the 9 nations along the shores of the Baltic Sea and the 6 nations bordering the Black Sea. Various efforts have also been undertaken to control nutrient inputs from other non-littoral nations lying within the catchments. The catchment area for the Black Sea, for example, covers nearly 1.7 million km^2 and includes some 17 nations and 160 million inhabitants. These efforts include multinational action plans to reduce nutrient sources in several large river basins, i.e., those of the Danube, Rhine and Elbe Rivers (Crouzet et al. 1999). In 1999 the Istanbul Commission for the Bucharest Convention in cooperation with the International Commission for the Protection of the Danube River recommended that all nations in the Black Sea basin should take measures to reduce the loads of nutrients to such levels necessary to permit Black Sea ecosystems to recover to conditions similar to those observed in the 1960s. These commissions agreed to an intermediate goal that control measures be taken by basin states to avoid discharges of nutrients into the Black Sea exceeding those which existed in 1997. Because of the collapse of the centrally planned economy in former Communist countries and resulting dramatic reduction in fertilizer use, the 1997 nutrient loading levels were approximately half of those in the late 1980s (Mee 2001).

The implementation of the commitments under the enclosed seas conventions is left to the signatory nations and the level of implementation varies greatly, especially considering the wide range in the economic status and governmental capacity of European nations. Directives adopted by the European Union (EU) that are to be implemented by nations have become a major force for reducing nutrient pollution because they are backed by the threat of sanctions. This is true even in Eastern Europe, where nations aspiring to EU membership are under pressure to comply with the directives as a potential condition for membership (Mee 2001). These include the Urban Waste Water Treatment Directive, which sets minimum standards for the collection, treatment, and disposal of wastewater dependent on the size of the discharge, and the type and sensitivity of the receiving waters; the Nitrate Directive, which aims to reduce or prevent the pollution of surface and ground waters caused by the application and storage of inorganic fertilizer and manure on farmland; the Directive on Integrated Pollution Prevention and Control, which is directed to major industries; and the Water Framework Directive, which calls for watershed-based approaches to improve water quality, specific goal setting and monitoring and stresses stakeholder involvement (Crouzet et al. 1999; Conley et al. 2002).

Driven by these multinational conventions and EU directives as well as their own domestic concerns, some European nations have implemented aggressive programs to reduce nutrient pollution to coastal waters. Notable among them is Denmark, which has extensive nutrient-impaired estuaries and coastal waters, intense agriculture, and is party to both the OSPAR and HELCOM conventions. In 1986 the Danish Parliament adopted an agenda for reducing total loads of N and P to the aquatic environment by 50% and 80%, respectively, and mandated action plans to achieve these objectives (Conley et al. 2002). To direct and track progress toward these objectives, the Danish government operates a substantial National Aquatic Monitoring and Assessment Program.

In the U.S. efforts to curtail deleterious nutrient inputs to coastal waters began as early as the 1960s with efforts to control runoff from duck farms into Moriches Bay on Long Island (Ryther and Dunstan 1971). Abatement of P over-enrichment in fresh waters had been underway for some time in bodies such as the Great Lakes (U.S. Environmental Protection Agency 1995) and P removal from municipal Washington, D.C. wastewaters began in the 1970s in an effort to improve water quality in the tidal Potomac River estuary (Jaworski 1990). Nutrient removal was incorporated in treatment of sewage discharges into Tampa Bay beginning in 1980 (Lewis et al. 1998) and eventually resulted in a 50% reduction in N inputs from these sources.

The first ambitious effort to effect major reductions of both point and diffuse sources of nutrients over a large region began with the 1987 agreement among Pennsylvania, Maryland, Virginia, the District of Columbia, and the federal government to reduce controllable inputs of both N and P into the Chesapeake Bay by 40% by the year 2000 (Boesch et al. 2001a). Controllable inputs were meant to exclude atmospheric sources, runoff from forests, and inputs from the three non-signatory states that occupy relatively small portions of the watershed. The Chesapeake Bay Program has developed and sustained extensive capabilities for assessment, including modeling and monitoring programs, and an expansive implementation structure that have been emulated for other U.S. coastal waters included under the National Estuary Program.

Of the 28 National Estuary Programs, 18 have identified the impacts of nutrient over-enrichment as a high or medium priority for management (Greening and Elfring 2002) and strategies to reduce nutrient loading are being implemented for 10 of these coastal ecosystems. Management programs for several heavily populated bays and estuaries with relatively small catchments rely heavily on reduction of point sources of nutrients, either through advanced waste treatment, such as in Tampa and Sarasota Bays and Long Island Sound, or through redirection of these discharges through deepwater outfalls, such as in Boston Harbor. A goal of reducing N loading by 58.5% has been set for Long Island Sound (Long Island Sound Study 1998). This is to be met mainly through advanced treatment of municipal and industrial wastes. For Tampa Bay because significant reductions in nutrient loading from sewage discharges have already been achieved, the goal is to hold gains made in nutrient load reductions by making additional reductions in point and diffuse sources to offset the effects of anticipated high rate of population growth in the region.

On a larger scale, in 2000 a task force, working under the auspices of the National Science and Technology Council that represented eight federal agencies, nine states, and two tribal governments, adopted a general goal of reducing the area experiencing hypoxia in the northern Gulf of Mexico by two-thirds (Rabalais et al. 2002). The task force recognized that this would probably require the reduction of N inputs from the Mississippi River Basin by 30%. As in the case of the Black Sea, achievement of this goal will depend heavily on actions taken within jurisdictions beyond the littoral states that are party to the regional sea compact which involves only five Gulf coastal states.

No specific nutrient source reductions are mandated under U.S. federal laws but two actions taken under the existing Clean Water Act (CWA) are destined to have an influence on nutrient inputs in U.S. coastal waters. One is the development of nutrient water quality standards by the U.S. Environmental Protection Agency (EPA). The other is the court-ordered implementation of a long-neglected provision of the CWA that requires that Total Maximum Daily Loads (TMDLs) of wastes be determined and allocated through regulatory action if technology-based standards do not succeed in achieving water quality standards for designated uses in water bodies (U.S. EPA 1999; NRC 2000, 2001).

Eutrophication of the coastal waters of Japan is a particular problem in the major urbanized embayments on the Pacific side of Honshu: Tokyo Bay, Ise Bay (including Mikawa Bay), and the Seto Inland Sea (including the heavily polluted Osaka Bay). In response to worsening water quality and red tides that interfered with aquaculture activities, the Law Concerning Special Measures for Conservation of the Environment of the Seto Inland Sea was enacted in 1978. Under the law, the 11 governors of the prefectures situated around the Seto Inland Sea established administrative guidelines for the reduction of loading of oxygen demanding materials and P by 1994. In 1994, N was added to this list. In 1993 the Japanese Environmental Agency established Environmental Quality Standards (EQS) and uniform national effluent standards for N and P to reduce marine eutrophication. These effluent standards are effective in the watershed areas of the 88 designated coastal waters. EQS for N and P have been applied to Tokyo, Ise, and Osaka bays and detailed target reduction figures are being set for 2004. Point sources in the form of domestic wastewater and industrial discharges dominate nutrient loads in these bays, although agricultural sources are also important in Ise Bay (Suzuki 2001)

Nutrient Reduction Goals

In the local to regional programs for reducing nutrient over-enrichment discussed here, the initial reduction goals were guided by a combination of professional judgment and political art. Literally within a few years (1986–1988) goals of 40% to 50% reduction in nutrient loadings were set for Denmark, OSPAR, the Chesapeake Bay, and HELCOM. Crude scientific models and nutrient budget reconstructions informed these decisions, but the narrow range of these initial goals suggests that political determination of targets that were significantly bold but acceptable by the public also played an important role. These goals were viewed as first steps that would achieve significant improvements. It was simply important to begin to reverse the trends.

Once the daunting proportions of these reductions were realized, it was deemed necessary to qualify what was included in the base to be reduced and what was not. For the Chesapeake Bay, the decision was made to exclude background loadings from forested landscapes, agricultural deposition, and loads from the three nonsignatory states falling partly in the watershed from the reduction targets. 40% reduction of the remaining controllable sources translated to reductions of 21% for total N loading and 36% for total P loading (Boesch et al. 2001a). During development of action plans based on the 1986 Danish legislation, overall reduction targets changed from being targets for the total loads to the aquatic environment to just reduction targets for the losses from agriculture, municipal waste treatment plants, and separate industrial discharges (Conley et al. 2002). Emissions of N to the atmosphere, storm water overflows, and P losses from agricultural fields were not included in the targets as implemented.

More recent nutrient reduction goals have been set based on simulation models or empirical evidence. The goal of reducing N loading by 58.5% for Long Island Sound was developed using hydrodynamic-water quality modeling to forecast hypoxic conditions. A level of reduced, but not eliminated, hypoxia was chosen as achieving an acceptable and attainable level of improvement (Long Island Sound Study 1998). The goal set for the northern Gulf of Mexico is also based on reduction of the extent of hypoxia, with a rough model approximation that this would require a 30% reduction of N loading (Rabalais et al. 2002). For the Black Sea and Tampa Bay the goal is to cap loadings at levels at which significant reductions of hypoxia, in the former case, and chlorophyll levels, in the latter case, had been realized.

Assessing Progress

Estimation of progress in meeting nutrient reduction goals is a challenge for several reasons. While many point sources can be monitored reasonably accurately, diffuse sources are harder to monitor. Nutrient delivery in stream flow may be highly variable on short time frames, posing estimation problems. Nutrient load is itself strongly related to stream flow, which varies greatly among years. Procedures for calculation of flow-adjusted loadings may be employed to normalize this variability (Cohn et al. 1989). Even then, there are time delays from when abatement actions are put in place to when these affect delivery of nutrients to coastal waters. These lags may be several years or more, particularly when significant groundwater flows are involved. Finally, it is difficult if not impossible to track these changes back to the particular abatement action (e.g., agricultural management practice or riparian buffer placement) involved.

Efforts to reduce nutrient loadings to coastal ecosystems are achieving some success. In Denmark, N loads from point sources were reduced by 66% and P loads by 81% during the 1990s (Conley et al. 2002). Although there were substantial reductions of total N and P loadings to Danish coastal waters, there was not a significant reduction of diffuse N loads, which mainly come from agriculture. Similar trends in inputs are evident elsewhere in Europe, with substantial declines in P loading due to reduced use of polyphosphate detergents and treatment of point sources and declines in N loadings where there has been advanced waste treatment of point sources (Crouzet et al. 1999). Diffuse sources of N, again mainly from agriculture, seem to have been little reduced. A notable exception is the decline in delivery of N and P to the Black Sea via the Danube River (and probably Dniester and Dnieper Rivers) five to seven years after the dramatic reduction in fertilizer use that occurred as a result of the economic changes of perestroika (Mee 2001).

Modeling procedures are being used to simulate progress as well as to forecast future consequences for reductions in nutrient loadings on watershed scales. Abatement actions are credited for certain reductions in nutrient loadings, and the changes in the loads delivered to the coastal water are estimated as a function of their location on the landscape and losses during delivery downstream. Such a hydrologic-watershed model is used in the Chesapeake Bay Program for both status assessment and forecasting (Linker et al. 1996). The model estimated that abatement actions were in place by the end of the benchmark year 2000 to result in effective reductions in controllable loadings of 34% for P and 28% for N (31% and 15% of the total loads, respectively; Boesch et al. 2001a). In addition to the assumptions regarding the credited loading reductions for the various abatement actions, this assessment normalizes flow conditions to the 1985 base year and includes no lag times for watershed delivery.

A more effective way to track progress in achieving goals for reduction of nutrient loading is to combine observations with models, allowing both the assimilation of observational data into the models and the identification of sources contributing to changes in observed loadings. Using such an approach, Behrendt (1999) found that N emissions into the North Sea from both the Rhine and Elbe Rivers declined by 29% during the 1990s, and that this was mainly due to substantial reductions of municipal and industrial point sources (46%). The decrease in diffuse inputs was only 10%, again pointing out the difficulties in reducing agricultural sources.

Assessing improvements in the quality of the receiving coastal waters provides the ultimate test of progress. There have yet been few unambiguous demonstrations of meeting objectives for coastal ecosystem rehabilitation from over-enrichment. The stepwise reduction of N loading to the Himmerfjärden, on the Baltic coast of Sweden, by 70% from 1976 to 1998 has resulted in decline in the concentrations of total N and chlorophyll *a* that approximated forecasts (Elmgren and Larsson 2001). Reductions in nuisance growth of macroalgae, increased water clarity, and reduced phytoplankton production have also been documented in Danish coastal waters (Conley et al. 2002). Chlorophyll *a* levels in Tampa Bay have been reduced to a level approximating management goals, but seagrass recovery has been slower than expected (NRC 2000).

In most other areas where nutrient inputs have been reduced, the trends are mixed or responses are delayed. In the Chesapeake Bay, statistically significant reductions in ambient nutrient concentrations have been observed in some regions where point source inputs have been reduced but not in the open bay or in areas receiving primarily diffuse source inputs (Boesch et al. 2001a). Trends in expansion of seagrasses or reduction of hypoxia are difficult to decipher from the effects of interannual variability of freshwater flows. Delayed responses due to nutrient storage in the watershed or coastal system and inherently nonlinear ecosystem responses should be expected. Positive feedbacks characterize some of these responses, potentially resulting in more dramatic improvements once thresholds are reached. For example, the gradual improvement in dissolved oxygen conditions should result in increased denitrification and P burial in bottom sediments (Boesch et al. 2001a). As the Black Sea experiment indicates, responses in the coastal ecosystem, in this case alleviation of hypoxia, may lag even dramatic reductions in nutrient inputs by as much as a decade (Mee 2001).

Second-generation Goals

While load reduction goals based on some practical approximation of outcome served a useful purpose of mobilizing efforts to reduce nutrient over-enrichment, new goals are being set to achieve more specific ecological conditions in the affected coastal waters. The objectives for Danish marine waters are that the fauna and flora may be affected only insignificantly or slightly by anthropogenic pollution, nutrient concentrations have to be at a natural level, the clarity of the water has to be normal, unnatural blooms of toxic planktonic algae or pollution-dependent macroalgae must not occur, and oxygen deficiency may only occur in areas where it is natural (Conley et al. 2002). The 2000 Chesapeake Bay Program Agreement does not set a specific load reduction goal, but specifies the objective of correcting nutrient-related problems needed to remove the bay and its tidal tributaries from the list of impaired waters under the CWA (Boesch et al. 2001a). Essentially this means determining TMDLs to meet water quality standards for designated uses. The process of determining designated uses and standards, estimating the maximum nutrient loads at which those standards can be met, and allocating the maximum acceptable loads among sources is underway not only for various segments of the Chesapeake Bay but also for many other rivers and coastal waters on the impaired list. Although not formally based on a TMDL, the agreement for N reduction in the Mississippi River basin also is based on an environmental endpoint—reducing the extent of hypoxia (Rabalais et al. 2002).

The EU's new Water Framework Directive sets up a remarkably parallel process to reconcile the inadequacies of the technology-based emission limits in achieving environmental quality standards. The Directive requires river basin scale approaches to defining good ecological status and the standards needed to achieve it, evaluation of the effectiveness of implementation of existing regulations to achieve these standards, and identification of additional measures needed to satisfy the objectives established. New Japanese policies require additional actions when effluent standards are insufficient to achieve environmental quality standards.

Such outcome-based approaches as U.S. TMDLs and the EU Water Framework Directive present major new challenges to science and the use of science (NRC 2001). Because returning to a pristine state, especially with respect to anthropogenic nutrient inputs, is seldom if ever a realistic option, outcome-based approaches involve the practical integration of what is desirable with what is achievable. Although in the end these are policy and not scientific determinations, science can and should play an integral role in informing the policymaking process both on desirability and achievability.

Abatement

Multipronged Approaches Typically Required

Some coastal waters (bay, estuary, or continental shelf region) receive nutrients predominantly from one source type (e.g., sewage discharges into Long Island Sound), such that a concentrated focus on abatement of nutrient loadings from that source (e.g., implementing advanced waste treatment) would be a sufficient strategy for reducing over-enrichment. Because most over-enriched coastal ecosystems receive loadings from numerous sources, an integrated strategy for effective abatement is required. The strategy should encompass the catchment basin, or watershed, draining into the coastal waters because of the widespread importance of diffuse sources. It may have to consider nutrients originating outside the catchment, but transported into it through the atmosphere. These are non-conventional units for ocean and coastal resource management and pose numerous challenges.

Through an integrated strategy substantial reduction in nutrient loadings to coastal waters may be achieved by approaches that reduce the use of the nutrients in the first place; control losses to the environment at the point of release (e.g., farm field, animal feeding operation, lawn or subdivision, vehicle, power plant, or industrial or sewage treatment works); and sequester or remove pollutants (enhance sinks) as they are transported to the sea. In order to achieve maximum effectiveness of these interventions at acceptable costs to society geographic targeting of source controls and sinks is important.

Sources

Significant reductions in P loading in Europe, North America, and Japan have resulted from discontinuing the use of polyphosphate detergents. P can be almost completely removed from wastewaters by additional chemical and biological treatment (NRC 2000). P removal from wastewater in the Washington, D.C. metropolitan area produced substantial improvements in water quality and living resources in the tidal freshwater portions of the Potomac estuary, which tend to be P-limited (Jaworski 1990). Advanced wastewater treatment for N removal has reduced point-source N loadings in Denmark and Sweden. Advanced treatment is being applied in Chesapeake, Tampa, and Sarasota Bays and Long Island Sound employing biological nutrient removal, a process that couples microbial nitrification under aerobic conditions with denitrification under anaerobic conditions (NRC 2000).

Reductions in nitrogen oxide (NO_x) emissions to the atmosphere have been driven by air quality considerations generally outside the influence of water quality or coastal ecosystem managers. In the U.S. NO_x emissions from power plants and vehicles are regulated under the Clean Air Act (CAA); a key goal of the 1990 Amendments to the act is to reduce ground-level ozone that poses human health risks and stresses forests and crops. Significant reductions in NO_x emissions from stationary and mobile sources are being mandated to meet CAA requirements (U.S. EPA 2000). A 40% reduction in NO_x emissions may ultimately be achieved as a result of new standards, technologies, and efficiencies being pursued under the CAA. In Europe it is estimated that a 15% reduction in NO_x emissions was achieved between 1990 and 1995, although this was largely due to declines in industrial activity in Eastern Europe (Crouzet et al. 1999). EU directives for reducing industrial emissions and exhaust emissions from on-road vehicles are expected to result in lowering NO_x releases by nearly one-third by 2010.

Abatement of agricultural sources of nutrient pollution is proving to be more difficult. To be practical, abatement of agricultural sources of nutrients must focus not only on reducing fertilizer use but also on plugging the many leaks in agricultural nutrient cycles. Although efficiencies in fertilizer use in U.S. agriculture have been slowly increasing since the mid-1970s, about one-third of the N applied is not recovered in harvested crops (NRC 2000). Not all of the missing N contributes to eutrophication of coastal waters. Much is denitrified in soils or aquatic systems en route to the sea or is stored in soils or groundwater. In addition to increasing the efficiency of N uptake by crops, the return of N_2 gas to the atmosphere can be enhanced through land management practices.

Various agricultural practices affect N and P runoff and losses to groundwater (which ultimately seeps into surface waters) or atmosphere. Practices employed to reduce soil erosion, such as contour plowing, timing of cultivation, conservation tillage (little or no tilling), stream bank protection, grazing management, and grassed waterways also reduce nutrient pollution. Other practices are more specifically targeted to the efficient use and retention of nutrients; including soil testing to precisely match fertilizer applications to crop nutritional needs (many farmers still overapply to insure maximum crop yields); applying fertilizer just at the time the crop needs it; crop rotation; planting cover crops in the fall; using soil and manure amendments, and specialized methods of application, such as injection (NRC 2000). Reducing the inefficiencies that generally lead to the application of fertilizers in excess of crop requirements can have significant benefits in terms of reducing nutrient loadings downstream. For example, using a hindcast, statistical model that adjusted for the effects of the annual water budget on nitrate flux in the lower Mississippi River, McIsaac et al. (2001) estimated that a 12% reduction in the application of fertilizers within the river basin could result in a reduction of 33% in nitrate flux to the Gulf of Mexico.

Landscape practices, such as maintaining buffer strips between cultivated fields and nearby streams, moderating excessive drainage by ditches and tile lines, and maintaining wooded riparian areas can further reduce the leakage of agricultural nutrients to surface waters. By combining these approaches a significant portion of the edge-of-field N losses can be reduced (Boesch and Brinsfield 2000; Mitsch et al. 2001).

Animal wastes are a significant source of nutrient pollution from agriculture. Although the total production of livestock has not dramatically increased in recent years, the number and size of concentrated animal feeding operations have. Volatilization of ammonia and its subsequent deposition on water bodies or their watersheds is an important source of N loading near areas of intense animal production (Paerl et al. 2002). Enclosures or trapping devices may eventually be required to stem ammonia emissions from animal wastes. Manure management also presents a risk of pollution if holding facilities fail or do not function properly. Because of the intensification of animal production, too much manure can be produced within a geographic area for it to be applied to nearby land without overloading soils with nutrients (NRC 2000; Sharpley 2000).

Urban runoff can also be an important diffuse source of nutrients. Reduction and control of urban and suburban diffuse sources can be achieved through reductions in fertilizer use; effective and well-maintained stormwater collection systems (retention ponds can remove 30–40% of the total N and 50–60% of the total P); and improved septic systems that promote denitrification (NRC 2000). Preservation and restoration of riparian zones and streams within urban and suburban areas is also an important aspect of effective nutrient control. The ability of streams to function effectively in nutrient removal is compromised when a significant portion of their watersheds is covered by impervious surfaces and the amplified runoff scours the stream beds (Booth and Jackson 1997).

Sinks

Removing or sequestering pollutants as they are transported downstream can also abate nutrient pollution. Many American and European watersheds were once sponge-like, containing extensive flood plains and wetlands that slowed the flow of water and served as sinks for dissolved and suspended nutrients through sedimentation, long-term storage in plant biomass or denitrification. Well over half of the wetlands present in the conterminous U.S. at the time of European settlement have now been converted to other land uses and the percentage of inland swamps and riparian wetlands lost is even greater (Mitsch and Gosselink 2000). Many floodplains have been disconnected from their rivers by flood control projects or agricultural conversion and no longer serve as nutrient sinks.

Reducing and controlling diffuse sources of land runoff must involve large-scale landscape management, including restoration of riparian zones and wetlands and management of lakes and reservoirs that serve as nutrient sinks. Mitsch et al. (2001) estimated that restoring 5 million acres of wetlands in the Mississippi River Basin would reduce N loading to the Gulf of Mexico by 20%. The Chesapeake Bay Program is striving to reforest 3,200 km of riparian zones and restore 10,000 ha of wetlands by 2010 in order to achieve nutrient reduction goals (Boesch et al. 2001a). The Danish Action Plan on the Aquatic Environment calls for re-establishment of 16,000 ha of wet meadows as nutrient sinks (Conley et al. 2002).

Nutrient sinks can also be enhanced within coastal ecosystems. Tidal wetlands serve as nutrient sinks, and facilitating the development of deltaic wetlands could remove a portion of the river-borne nutrients discharged to coastal waters in regions such as the Mississippi delta (Mitsch et al. 2001). The Chesapeake Bay Program has a goal of a 10-fold increase in oyster biomass in order to recover the natural biofiltration capacity (Boesch et al. 2001a). Biodeposits associated with oyster reefs exhibit very high rates of denitrification (Newell et al. In press). Rebuilding tidal flats that have been lost and degraded as a result of the extensive coastal development within Japanese bays is estimated to be an important purifying mechanism to reverse eutrophication of those ecosystems (Susuki 2001)

Targeting

Geographically targeting source controls and riparian and wetland restoration is critical to the effectiveness of these efforts in controlling nutrient loadings downstream. Statistical models based on water quality measurements throughout the Mississippi River Basin show that the percentage of N leached from a field that reaches the Gulf of Mexico depends greatly on its proximity to larger streams and rivers (Alexander et al. 2000). Biological uptake and denitrification are already effective in small watercourses, particularly if they are in healthy condition (Peterson et al. 2001). Agricultural source controls should be directed in particular to those lands that are disproportionately responsible for loads delivered to coastal waters. Restoration of riparian and wetland habitats along moderate to large streams should also be more cost-effective. Because of equity considerations, however, both incentives (subsidies and cost sharing, technical assistance, and insurance) and disincentives (regulatory controls, taxes, and fees) for abatement tend to be applied relatively uniformly.

Challenges and Opportunities for Science

With the turn of the century, the science of eutrophication faces very different challenges and opportunities. The emphasis during the late 20th century was documentation of the degradation of coastal ecosystems and diagnosis of its root causes. With the scope, consequences, and causes of coastal eutrophication now largely well documented (although this is clearly less the case in the developing world), science must transition to a preventative and rehabilitative stage, with an emphasis on prognosis and treatment. Difficult challenges as well as exciting opportunities face the scientific community and environmental managers in determining susceptibility, defining outcomes that are both achievable and desirable, reducing sources, enhancing sinks, targeting efforts for optimum effect, and forecasting and observing responses.

Determine Susceptibility

Improved understanding of the degree and nature of susceptibility of coastal ecosystems to nutrient over-enrichment is required not only for targeting efforts to protect threatened systems (NRC 2000; Greening and Elfring 2002), but also for managing the rehabilitation of ecosystems degraded by multiple stresses. Three of the five questions posed in Cloern's (2001) review of the science of coastal eutrophication are directly relevant to the objective of determining susceptibility: How does the coastal ecosystem filter work? How does nutrient enrichment interact with other stressors? How are responses to multiple stressors linked? These multiple stressors include translocation of species, habitat loss, fishing, inputs of toxic contaminants, manipulation of freshwater flows, aquaculture, and climate change.

Comparative analyses of the responses of coastal ecosystems to nutrient enrichment are emerging, but have not yet developed a comprehensive understanding of these responses on a global scale (Cloern 2001). Massive amounts of information are accruing from monitoring and research programs that should make such a grand synthesis possible. From a practical standpoint, this is necessary to guide protection and rehabilitation efforts to where they are most needed and to avoid costs of controls where they have little result. Approaches described in the NRC (2000) assessment for classifying estuaries and bays according the dilution and flushing by freshwater inflow and tidal prism volumes to yield susceptibility indicators offer a good first step.

Define Desirable and Achievable Outcomes

Goals or standards for ecosystem protection or rehabilitation represent compromises based on what is desirable and what is achievable. Cloern's (2001) fourth question is germane to determining desirable states for the ecosystem: How does coastal eutrophication impact the Earth system as habitat for humanity? Answering this question requires knowledge of the consequences at higher trophic levels (e.g., fisheries) of direct importance to humans, but also the consequences to ecosystem services, the global climate system, and human health and culture. Science should also add realism in the definition of achievable outcomes and the constraints, costs, and choices implicitly embedded therein. Science needs to remind management and policymaking that ecosystems are dynamic and variable and are subject to long-term changes such as estuarine infilling, deltaic senescence, and climate change. The potential consequences of 21st century climate change on eutrophication and outcome goals (Justić et al. 1996; Najjar et al. 2000) should be brought into consideration. Outcome goals need to take into account the variability and secular changes that may take place within and beyond environmental management horizons.

Reduce Sources

Grappling with the challenges of reducing diffuse nutrient sources, tracking the downstream fate of nutrients, and understanding coastal ecosystem responses presents an exciting opportunity for the development of holistic environmental science that transcends atmospheric, terrestrial, aquatic, estuarine, and marine environments. Except for biogeochemists and engineers who develop transmedia budgets and models, few scientists seem prepared to delve beyond the boundaries of their medium-centric disciplines. This is unfortunate because important insights can be developed and contributions made by those prepared to make the trip. This is particularly the case for the development of effective and efficient source controls, where the agricultural scientist may develop new ideas for source controls by understanding downstream fate and coastal responses or the coastal ecologist may contribute important insights on the effectiveness of various nutrient controls or interactions among N, P, and silica. Science and technology to improve the effectiveness of nutrient source controls, both from diffuse sources and in advanced waste treatment, must be a high priority in the preventative and rehabilitative stage of eutrophication assessment and management. This requires not only intensive agronomic research, for example, but also placing this research into a broader landscape and watershed context.

Enhance Sinks

Science and ecotechnology related to enhancing nutrient sinks must also be a high priority in the prognosis and treatment of eutrophication. As with source controls, they should also be conducted within a landscape and seascape context. Key issues include the strategic siting of ponds, wetlands, and riparian buffers within the landscape and along water courses (Mitsch et al. 2001); their operation and maintenance; and effective design of coastal and estuarine habitat restoration to enhance coastal nutrient sinks such as salt marshes, tidal flats, seagrass beds, and oyster reefs.

Target Efforts

Geographic information systems and spatially explicit models enhance the ability for targeting of source reduction and sink enhancement efforts. SPARROW (Spatially Referenced Regressions on Watersheds) modeling based on observed water quality data provides an empirical, strategic tool for this on regional scales (Smith et al. 1997). Mechanistic watershed models of process rates and states allow one to estimate delivered nutrient loads resulting from source reductions on progressively finer special scales, but, of course, are sensitive to model assumptions (NRC 2000). Spatially explicit process models can even incorporate economic drivers and costs (Costanza et al. 2002). Research on source controls and sink enhancement should be more effectively linked with such modeling approaches. Management intervention, for example in the placement of cropping systems including cover crops, manure applications, and reconstructed wetlands and riparian buffers, could be made much more effective and less costly. Concerns regarding equity and regulatory uniformity tend to work against such geographic targeting; these technical analyses contribute to assessments of optimum social benefits and costs.

Predict and Observe Responses

Efforts to reduce nutrient over-enrichment of coastal ecosystems must place a premium on environmental modeling and monitoring (NRC 2000), particularly if they involve controlling multiple sources within a watershed (NRC 1999). Forecast models are needed not only to target abatement, but also to track the consequences of source control and sink enhancement that propagate throughout the watershed and to project coastal ecosystem responses. Monitoring observations are critical to determining the effectiveness of abatement strategies, evaluating outcomes with respect to goals, and placing the responses into the context of ecosystem variability.

To be effective in an adaptive management framework, modeling and monitoring should be intimately coupled but seldom are. Models of environmental processes as complex as watershed transport and nutrient and community dynamics within coastal ecosystems can be simulated only with significant uncertainty (Cloern 2001). Overreliance on single, cross-scale models of biophysical processes was identified by Walters (1997) as one of the principal impediments to the implementation of adaptive management. Focusing exclusively on monitoring results without incorporating predictive modeling has been a criticism leveled at the Danish monitoring and assessment program (Conley et al. 2002). The pursuit of outcome-based, TMDL-like strategies for reducing nutrient over-enrichment to desirable and achievable levels requires a variety of modeling approaches—simple and complex, statistical and mechanistic—integrated with monitoring observations in the adaptive implementation of management programs (NRC 2001).

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