Coastal Eutrophication and Agriculture: Contributions and Solutions

DF. Boesch[1], RB. Brinsfield[2]

Many coastal waters of developed nations have experienced widespread and rapid eutrophication (the increase in supply of organic matter) during the last half of the 20th century. This has resulted in increased phytoplankton production, decreased water clarity, often-severe depletion of dissolved oxygen in bottom waters, loss of seagrasses and, in some cases, declines or changes in the quality of fisheries production. Large-scale changes resulting from eutrophication have been documented for continental shelf waters in the Gulf of Mexico, Mediterranean, Black and North Seas, relatively confined seas such as the Baltic and Seto Inland Sea, large bays such as the Chesapeake Bay and Long Island Sound and numerous smaller estuaries and lagoons. These trends are closely tied to the increased use of chemical fertilizers in agriculture, human population growth, and increasing atmospheric deposition of nitrogen resulting from fossil fuel combustion. Although atmospheric and human waste sources are significant in some heavily populated areas, agricultural inputs of phosphorus and nitrogen are the largest source of nutrients driving the increased production of organic matter in most extensively affected areas, including coastal waters receiving drainage from large river basins with extensive agriculture (e.g., Mississippi, Po, and Danube Rivers).

Agricultural inputs of nutrients are driven not only by applications of chemical fertilizers, but also by animal wastes, irrigation, drainage, and the conversion of wetlands and riparian zones (important sinks for nutrients) for agricultural land uses. More efficient agronomic practices, use of crop rotation and cover crops, and avoiding the over-application of manure can result in reductions in nutrient losses by 20 to 30%. Reconfiguration of agricultural landscapes through reconstruction of strategically placed wetlands, riparian forests and flood plains can trap a similar fraction of the remaining nutrient losses, such that total reductions of 50% may be feasible without devastating economic impacts and with numerous local benefits to environmental quality. Efforts to restore large coastal ecosystems such as the Baltic Sea,

---

1. University Center for Environmental Science, P.O. Box 775, Cambridge, Maryland 21613, USA
2. Wye Research and Education Center, College of Agriculture and Natural Resources, University of Maryland, College Park, Queenstown, Maryland 21658, USA

In: Balázs, E., E. Galante, J.M. Lynch, J.S. Schepers, J.-P. Toutant, D. Werner, and P.A.Th.J. Werry (eds). 2000. *Biological Resource Management: Connecting Science and Policy*. Springer, Berlin.

northern Gulf of Mexico, and Chesapeake Bay through commitments to reduce nutrient loading have been underway or are beginning. They represent substantial challenges in working across political jurisdictions and across scientific disciplines and internalizing the external environmental costs of food production.

Introduction

Eutrophication – the increasing supply of organic matter (Nixon 1985) – is one of the most serious threats to coastal marine ecosystems around the world (GESAMP 1990; NRC 1994). However, the most substantial and widespread eutrophication has resulted from increased delivery of plant nutrients, not organic wastes, to coastal waters as a result of human activities. This differs from the typical lay person's notion of pollution, namely the direct discharge of a toxic chemical. On one hand, inputs of nitrogen, phosphorus, and other nutrients are essential to their high productivity. On the other hand, oversupply by nutrients may cause excessive algal growth. Such over enrichment results in diminished light penetration and shading of bottom-dwelling plants, harmful or noxious blooms of algae, altered food chains, and depletion of the dissolved oxygen as the organic matter decomposes.

Agriculture has played an important role in the delivery of nutrients to coastal waters since the land was first cleared and cultivated. Virtually any land-use activity adds nutrient; changes the capacity of the native plants, microbes, and soils to retain nutrients; and increases water and sediment runoff. However, the increased use of chemical fertilizers and nitrogen-fixing crops, the expansion of areas in cultivation, higher yields, and intensification of animal production have greatly increased nutrient loss from agricultural activities into streams, rivers, and eventually the sea.

Coastal eutrophication has grown to such proportions and severity that coastal constituencies and governments are demanding control on the sources of nutrient inputs, not only within the coastal zone but far upstream. Various national and international agencies are aggressively promoting voluntary practices for reduction of nonpoint sources, including those in agriculture, and, in some cases, implementing mandatory requirements. At the same time, agriculture is facing the challenge of feeding a growing world population, the economic pressures of new global markets, and competition with growing cities for productive land.

This chapter provides an overview perspective of the current state of knowledge regarding the causes and consequences of coastal eutrophication, the role of agriculture, and the new approaches that could be taken to reduce nutrient losses from agriculture. We also discuss efforts underway to control nutrient inputs from large, multijurisdictional watersheds and the institutional and scientific challenges they present.

Coastal Eutrophication: A Late 20th Century Phenomenon

Cultural Eutrophication

The environmental effects on coastal ecosystems of the addition of large quantities of organic matter from sewage or industrial discharges such as pulp plants have been recognized and studied for some time. Such concentrated inputs of organic matter can result in depletion of dissolved oxygen due to the biochemical oxygen demand of the wastes and often cause a buildup of organic matter in sediments.

dramatically affecting bottom-dwelling (benthic) organisms (Pearson and Rosenberg 1978). While these effects may be severe and require long-term recovery, they are generally of limited spatial extent.

While the effects of inorganic nutrients on eutrophication of lakes have been recognized and studied for some time, recognition of the extent of coastal eutrophication from land-based nutrient inputs and research on the subject is relatively recent, with rapid growth in the numbers of papers published only since 1985 (Nixon 1995). There are three explanations for this. First, the confined nature of lakes and low turnover of their water volume makes them particularly susceptible to nutrient eutrophication (Smith 1998). Second, the loading of nutrients to the coastal zone has dramatically increased since the 1950s and 1960s (Howarth et al. 1996). Finally, scientists have only recently developed the tools to distinguish the effects of eutrophication in highly variable coastal ecosystems and to reconstruct historical changes.

The effects of overenrichment by nutrients are more insidious than those due to organic discharges. They are not concentrated at the point of input, but are spread over broader areas as nutrients stimulate the production of particulate organic material that is subsequently transported and mineralized. Moreover, an atom of carbon contained in organic matter is practically lost from the system once it is oxidized, while an atom of nitrogen or phosphorus may be recycled several times during residence in the coastal ecosystem, each time facilitating the production of more organic carbon (Ryther and Dunstan 1971).

The delivery of nitrogen and phosphorus to coastal waters has increased multifold due to human activities during the 20th century. In creating N and P budgets for the North Atlantic Ocean, Howarth et al. (1996) estimated that nitrogen fluxes from the continents had increased from 2 to 20 times from the preindustrial period. The increase in N and P delivery to coastal waters can be attributed to the growth, concentration, and consumption of human populations and the dramatic increase in the use of chemical fertilizers and combustion of fossil fuels during the last half of the century. Fossil fuel combustion results in the release of oxides of nitrogen to the atmosphere, which return to the Earth's surface in acidic wet deposition or adsorbed onto particles. Agricultural and atmospheric deposition sources vary among regions of the North Atlantic basin as a result of differences in population density, fuel consumption, and agricultural activity.

The extent and consequences of eutrophication in the developing world are less well understood. Coastal ecosystems in these regions have been as extensively studied as those in developed regions. Nutrient limitation and the effects of eutrophication in tropical waters are, in particular, poorly known. Nutrient inputs per capita are lower in the developing world because of the lower consumption of fertilizers and fuels. However, as population growth takes place primarily in the developing world, urbanization intensifies, and consumption of fertilizers and fuels grows to support demands of development, serious risks are posed for the worsening eutrophication outside of North America, Europe, and Japan (Nixon 1995).

Although phosphorus and other micronutrients may stimulate algal production in coastal waters, particularly in brackish waters, nitrogen is the limiting nutrient for most coastal marine ecosystems. Algal production in marine ecosystems has been shown to be nitrogen-limited because nitrogen is continuously lost to the atmosphere due to denitrification, while phosphorus is relatively conserved. Moreover, cases cover thousands of square kilometers, with the most extensive hypoxia found in the Baltic, Black and Adriatic Seas and the northern Gulf of Mexico (Diaz and Rosenberg 1995). These regions receive runoff from large rivers draining the continents and have modest tides for mixing waters. As a result, the water masses are often seasonally stratified, restricting the resupply of oxygen to cooler and saltier bottom waters sufficient to meet the demands of decomposition of organic matter from the nutrient-enriched surface waters, causing depletion or elimination of dissolved oxygen in bottom waters.

Evidence is also building up in many affected areas that eutrophication has increased dramatically in the 20th century. The best records are for European waters such as the Baltic Sea (Larsson et al. 1985; Jansson 1997; Jansson and Dahlberg 1999), the Kattegat (Jørgensen and Richardson 1996), and the Adriatic Sea (Malone et al. 1999), where there is a long history of scientific observation to document the changes taking place. For example, in the northern Adriatic Sea analysis of oxygen data beginning in 1911 shows there was a significant shift toward supersaturated oxygen conditions in surface waters (reflecting high rates of photosynthesis) and hypoxic bottom waters between 1966 and 1972 (Justić et al. 1987; Justić 1991). This was coincident with more than doubling of the average concentration of N and P in the discharge of the Po River, the largest source of land-based inputs (Justić et al. 1994). In the Baltic Sea declining oxygen concentrations in the deep basins (for which records began in the 1930s) were observed beginning in the 1960s and continued to the present (Diaz and Rosenberg 1995), while in the Kattegat (for which observations began in 1912) worsening hypoxia and progressive mortalities of benthic organisms began in 1980 (Jørgensen and Richardson 1996).

In other parts of the world, including North America, the history of careful scientific observation is not as long. However, the use of biological and biogeochemical paleoindicators in sediments allows the reconstruction of the history of eutrophication. In the Chesapeake Bay – the largest estuary in the United States (4400 km^2) into which drains a 166 000 km^2 watershed – records of sedimentation, diatoms and pollen chronicle the first signs of eutrophication associated with land clearing by European colonists, which brought increased loads of sediments and nutrients to the estuary beginning in the late 18th century (Cooper and Brush 1991; Cooper 1995). Based on geochemical indicators, periodic hypoxia began to occur in the deep waters of the bay; however, seasonal anoxia intensified during the 1950 to 1970 time period. On the continental shelf off the mouth of the Mississippi and Atchafalaya Rivers, the first systematic observations of hypoxia in bottom waters began in 1985 and continued to document the occurrence of hypoxia over approximately half of the year and over an area from 9000 to 18000 km^2 in mid-summer. Although there is some suggestion in the paleoindicators of a signal of land clearing in the Mississippi basin in the 19th century, the most dramatic changes in planktonic production and hypoxic stress on benthos were observed during the late 1950s and 1960s (Rabalais et al. 1996).

Anoxia and hypoxia are extreme modifications resulting from eutrophication. Other effects may precede serious hypoxia. Moreover, increased algal biomass, decreased light penetration and altered benthic habitats may result in smaller coastal bays and estuaries not susceptible to hypoxia because of the lack of water mass stratification. While the timing of changes noted in these different parts of the world varies somewhat, it is clear that significantly accelerated eutrophication took place during the 1960s and 1970s. For example, algal blooms and hypoxia on the northwestern shelf of the Black Sea expanded dramatically (Tolmazin 1985), while nitrate and phosphate concentrations in the Danube River increased by up to tenfold (van Bennekom ans Salomons 1981) and nutrient concentrations in the Denstr River doubled (Balkas et al. 1990). Increases in nutrient loading were coincident with the rapid increase in fertilizer use and crop production in North America and Europe. In fact, Turner and Rabalais (1991) demonstrated the close coherence between the concentration of nitrate in the Mississippi River discharge, which nearly tripled (Goolsby et al. 1999), and the use of chemical fertilizers in the United States. However, the role of agricultural activities in coastal eutrophication is complex and will be explored later.

Effects of Eutrophication

While the processes resulting in oxygen depletion of bottom waters from eutrophication are now reasonably well understood, the consequences of the these and other ecosystem changes induced to fisheries production and overall health of coastal ecosystems are poorly known.

Nutrient inputs to coastal ecosystems, whether from runoff or upwelling from the deep ocean, are essential to maintaining these highly productive ecosystems. Primary production in marine ecosystems is directly proportional to the supply of nutrients (Nixon et al. 1986); consequently, increase in the nutrient supply invariably leads to increase in plant production somewhere in the receiving system, particularly by phytoplankton. For example, Richardson and Heilmann (1995) calculated that annual primary production in the southern Kattegat had increased by at least a factor of two between the periods 1954-1960 (when ^{14}C methods of measuring production were first used) and 1984-1993. While some of this increased production is consumed by zooplankton grazers and may thus support higher trophic level production, under eutrophic conditions a larger proportion of the organic production typically sinks (passively or through biodeposition) to bottom waters and the seabed, where its decomposition consumes oxygen.

In addition to affecting the rate of plant production, nutrient enrichment also affects the standing crop biomass (and thus light penetration) and the qualitative composition of the phytoplankton. The sediment record in the Chesapeake Bay shows an increase in small planktonic diatoms and decrease in benthic diatoms as the Bay transitioned from a clear water system with extensive benthic primary production to a more turbid system dominated by smaller phytoplankton (Cooper 1995). Such shifts to smaller, fast-growing phytoplankters (particularly flagellates and nannophytoplankton) under eutrophic conditions are thought to alter planktonic food webs in favor of small protist consumers and microbial recyclers – the so-called microbial loop. This not only shunts production from the food chains supporting fish larvae and other higher consumers, but results in more rapid recycling of nutrients, which can then stimulate more production.

Some of the phytoplankters favored by eutrophication bloom to nuisance proportions or produce toxins that kill other organisms and may even affect human health. For example, in the Neuse River estuary, a part of the extensive, shallow Albemarle-Pamlico system in North Carolina, USA, large agricultural and other inputs of nutrients stimulate blooms of cyanobacteria and dinoflagellates that cause water discoloration and nuisance conditions as well as depletion of oxygen (Pinckney et al. 1998). Nutrient enrichment has also been suggested to contribute to outbreaks of the mainly heterotrophic dinoflagellate *Pfiesteria piscicida* (Burkholder and Glasgow 1997), an organism that produces toxins that can kill fish and result in impairment of short-term memory and other cognitive functions in humans (Grattan et al. 1998). Because of risks to human health, these and other outbreaks of toxic *Pfiesteria* along the US east coast have raised public fear and stimulated regulatory and legislative action to control nutrient inputs to estuarine waters. There is international concern about the apparent increased frequency of a variety of harmful algal blooms around the world (Anderson 1995). While it is not clear that eutrophication is responsible for this general increase in harmful algal blooms – and certain that it is not for some types of algae – some toxic blooms in European and Japanese waters have been linked with nutrient overenrichment (Smayda and Shimizu 1991). Much yet needs to be learned regarding the relationships between harmful algae and nutrient enrichment. For example, the form of nutrients (e.g., organic N versus nitrate) may be important. Already, though, concern about risks to human health is a new driving force shaping policies for the control of nutrient pollution both in North America and Europe.

Organisms living at the seabed may be affected by eutrophication in several ways. Increased nutrient availability may favor certain epibenthic algae over others. In the Baltic, for example, annual filamentous algae are favored over coarse perennial algal species (Jansson 1997). Shading by dense phytoplankton and epiphytic growth may keep sufficient light from reaching seagrasses (Duarte 1995) or corals causing their demise. Increased organic matter deposited on the seabed provides an enriched food resource for some deposit feeders living in the sediments, while increasing physiological stress (due to low oxygen or high sulfide levels) in others. As a result, the density, biomass, and composition of benthic communities are frequently greatly altered (Diaz and Rosenberg 1995). Finally, hypoxia or anoxia may extend into the water column, stressing or killing animals living on or swimming near the bottom. In addition to influencing the abundance and composition of the benthic biota, eutrophication may affect rates of sedimentation, decomposition of detrital material, bioturbation of sediments, nutrient regeneration and material exchange across the sediment-water interface (Heip 1995).

Experience from several regions of the world indicates that modest eutrophication can increase fishery harvests, particularly of pelagic species, as a result of food chain stimulation (Caddy 1993; Houde and Rutherford 1993). On the other hand, when seasonal or permanent hypoxia begins to occur, catches of benthic feeding fish decline rapidly. If oxygen deficiencies and other manifestations of hypereutrophy continue, even pelagic populations may be adversely affected. Perhaps the most extreme case of devastation of fisheries by eutrophication is the northwestern Black Sea, where numerous stocks of benthic and pelagic species have collapsed (Mee 1992; Caddy 1993). Of the 26 commercial species fished in the 1960s, only 6 still support a fishery.

The interactions between fishing pressures and eutrophication make the relationship of enrichment to fishery yields difficult to interpret. Fishing effort and, consequently, catches have in some cases increased along with eutrophication, perhaps giving the false impression that fish populations have increased or at least not been affected by nutrient enrichment. On the other hand, in some areas, overfishing has reduced stocks while eutrophication proceeded, potentially lending the false impression that nutrient over-enrichment was responsible.

Finally, eutrophication may have manifold effects on the overall "health" of a coastal ecosystem in the sense of its vigor, organization, and resilience (Costanza 1992). Vigor embodies the throughput or productivity of an ecosystem. Organization represents not only its species diversity, but also the degree of connectedness of the constituent species. Resilience refers to an ecosystem's ability to maintain structure and patterns of behavior in the face of disturbance. A healthy ecosystem, then, is one that is active, maintains its biological organization over time and is resilient to stress. The gross productivity of the Chesapeake Bay, for example, has increased as a result of two centuries of cultural eutrophication (Boesch, et al. 1999), but this increase results from rapidly growing, small organisms at the expense of larger, more long-lived organisms that are more valuable to humans. From this perspective, the health of the Chesapeake Bay has deteriorated as a result of nutrient overenrichment, concomitant reduction of light availability, and loss of habitats that produce complexity. This has resulted in an ecosystem that is a less vigorous producer of valuable fish and shellfish, less diverse and well organized, and more susceptible to and slower to recover from disturbance (Ulanowicz, 1997).

Contributions from Agriculture

Nutrient delivery to coastal waters is affected by agricultural activities through a number of interacting activities. Most attention has been given to the expanded use of inorganic fertilizers in agriculture, for indeed it is this form of anthrogenically fixed nitrogen that has grown most dramatically since the 1960s (Vitousek et al. 1997). While the use of industrially fixed nitrogen is a key driving force for agricultural inputs of nitrogen to coastal waters, the sources and pathways are diverse and interacting and, thus, must be considered in control strategies.

Chemical fertilizer use in Europe and North America grew rapidly after the 1950s, but has leveled off since the 1980s. Worldwide, however, fertilizer use has continued to grow in meet increased demand in developing countries (Nixon 1995). Nitrogen and phosphorus in chemical fertilizers applied to fields may be transported to coastal waters in a number of ways, including movement of unassimilated materials into groundwater or surface waters, transport in the air of volatilized (NH_3) or dust-associated materials, and the mineralization of plant residues. Nitrogen and phosphorus behave differently in these regards, with nitrogen as nitrate being highly soluble and prone to groundwater intrusion, and phosphorus being less soluble and highly particle-reactive. The proportion of the N or P applied to fields that is lost to aquatic systems depends on the degree to which the application rate meets or exceeds the crop's requirements (overapplication results in proportionately greater losses), tillage practices, soil type and underlying geology, drainage, and climate conditions.

Although excluded from some budgets of new nitrogen inputs and agricultural source controls, nitrogen-fixing crops contribute nitrogen to terrestrial ecosystems far beyond the rate of nitrogen fixation in forests and grasslands. Jordan and Weller (1996) estimated that new N inputs by nitrogen-fixing crops such as soybeans and alfalfa are more than two-thirds of that due to inorganic fertilizers for the entire United States. The decomposition of the unharvested plants releases nitrogen, which may be lost to surface or groundwaters (Angle 1990), although Goolsby et al. (1999) concluded that the extensive soybean cultivation in the Mississippi River basin contributed little of the nitrogen exported by the river.

Considerable losses of N and P to coastal waters may result from the handling, storage and application of animal wastes to agricultural land. The trend of intensification of poultry and livestock production concentrates animal wastes, thus increasing the likelihood of these losses. Nutrients may be lost due to volatilization of wastes, discharge from waste holding facilities, and the application of manure as a soil fertilizer.

Atmospheric return of NH_3 volatilized from fertilizers or animal wastes is an important route for N input into coastal waters in some regions. For example, atmospheric deposition has been shown to account for over 30% of the nitrogen supplied to North Carolina estuarine waters (Paerl 1995) and is similarly important in regions of the Baltic and North sea, including The Netherlands, Belgium, and Denmark (Asman and Larsen 1996). Monitoring of waste streams and receiving waters may underestimate the importance of atmospheric transport routes of N delivery.

Artificial drainage of agricultural lands and alterations of the hydrology and plant communities of field-edge and downstream riparian zones may also greatly increase the delivery of nutrients to the coast. Soils in many highly fertile farmlands, such as the corn belt of the upper Mississippi River valley or low coastal lands or flood plains of the are naturally hydric and are able to grow crops only because they are artificially drained. This has three major effects on nitrogen export: (a) large reservoirs of organic matter in the soil are oxidized, releasing nutrients through mineralization; (b) soluble nitrate is more readily leached into drainage tiles or ditches; and (c) denitrification, which occurs in anoxic, wet soils, is reduced. In fields in the upper Mississippi basin that have intense tile drainage loss of nitrogen from mineralizing soil organics may be greater than that from chemical fertilizers (Cambardela et al. 1999).

The complex sources, sinks, and transport and transformation processes operating on nutrients make it difficult to account for the origins and causes of coastal eutrophication downstream. Through a variety of mass balance and statistical modeling approaches, Goolsby et al. (1999) have developed a nitrogen mass balance for the Mississippi River Basin. Fertilizer accounts for 54% of the new nitrogen inputs, N fixation 34%, and atmospheric deposition of nitrate (from fossil fuel combustion) the remainder. Recycled nitrogen (from people, animals, the atmosphere, and mineralized soil) is estimated to equal nearly 85% of the new nitrogen inputs, with inputs of N from soil mineralization about equal to that from fertilizer. Discharge from the river into the Gulf of Mexico accounts for only 7% of the total nutrient budget (or 13% of the new nutrients), slightly less than the amount of nitrogen lost to the atmosphere due to denitrification. In aggregate, though, agricultural sources account for an overwhelming proportion of the nitrogen budget of the Mississippi basin (Figure 2).

The relative contributions of agricultural sources to nutrient delivery to coastal waters vary widely, however, among the regions experiencing eutrophication (Fig. 2). The Mississippi River basin is heavily dominated by agricultural sources. Municipal and industrial point sources contribute only a very small percent of the N or P that reaches the Gulf of Mexico. Although agricultural sources are significant both for the Chesapeake Bay (with its grain fields and meadows supplying intense animal production), the northern Adriatic (which receives the effluent from the agricultural Po River valley) and Denmark, their more concentrated human populations contribute relatively more wastes via point sources and combustion by-products. Moreover, some eutrophied coastal waters, such as Long Island Sound (New York) and the Seto Inland Sea (Japan), receive little input of nutrients from agriculture but large inputs from outfalls and the atmosphere from nearby major population centers.

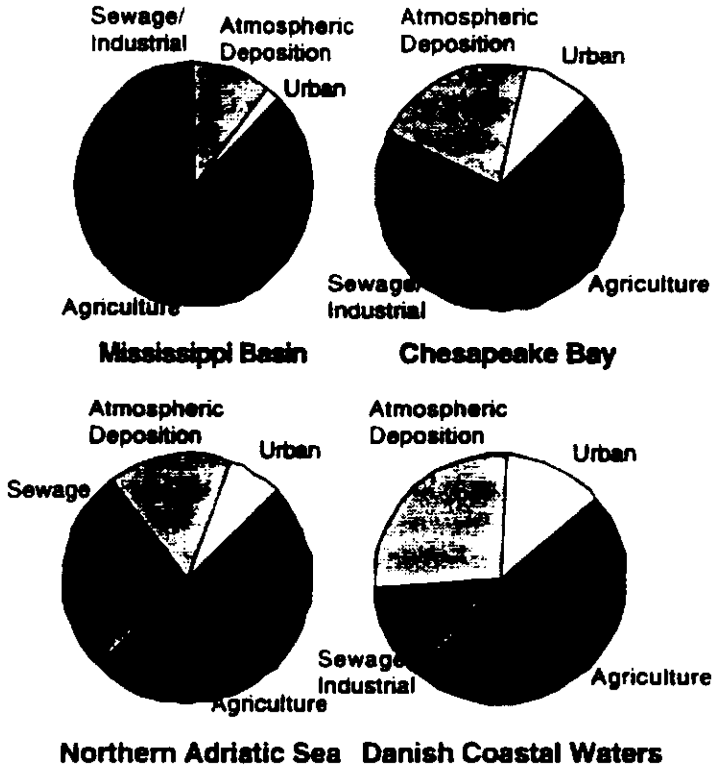

Fig. 2: The relative contribution of agricultural sources in comparison to other anthropogenic nitrogen loads varies among eutrophic coastal regions (Boesch et al., 1999; Goolsby et al. 1999; Seagle et al. 1999; Kronvang et al. 1993)

Approaches to Reduce Nutrient Losses

Introduction

Mitigating the environmental impacts of food and fiber production on our natural resources represents one of the greatest challenges facing agriculture as we enter the next millennium. Finding solutions that are environmentally sound and economically feasible heighten the challenge. Presently, these challenges are particularly acute for the livestock and poultry industries. The general trend from small-scale, diverse family farms to large integrated containment facilities located on limited land area further exacerbates these problems and makes economically viable solutions ever more challenging. Also challenging is the management of nitrogen losses from extensive grain production resulting from artificial drainage or seepage into groundwaters. Control of these losses requires not only more efficient application of fertilizers, but also the large-scale restoration of wet landscapes that act as sinks for nutrients.

Due to differing mobility in the soil, groundwater, and surface waters, approaches to the reduction of nitrogen and phosphorus losses are considered separately below. N controls focus on the movement of nitrate (NO_3) through the soil profile to groundwater, while P generally moves as either soluble or particulate P in surface water runoff.

Nitrogen

Nitrate contamination of groundwater has been documented in most regions of the United States and Europe, where N fertilizers, including sewage sludge and animal manure, are applied. Elevated N in groundwater not only poses a human health concern due to contaminated drinking water, but also contaminates surface waters due to lateral transport of groundwater into streams.

Most of the regions that have experienced increasing delivery of nitrogen to coastal waters are associated with shallow aquifers, coarse-textured soils, excessive manure-N application rates, dense animal populations, overapplication of fertilizer, highly mineralizing soils, or hydrologic conditions conducive to leaching, e.g., excess rainfall, heavy irrigation, artificial drainage, limited crop uptake, and limited denitrification losses (Sharpley et al. 1997).

With increasing attention to protection of groundwater, future attempts to improve water quality should focus first on source control; specifically on preventing nitrate from entering subsurface flow paths. Developing strategies to reduce nitrate loss to groundwater requires information about the temporal and spatial patterns of nitrate leaching in agricultural systems. Rates of N leaching have been correlated with N fertilization rates in various systems (Baker and Johnson 1981; Mitsch et al. 1999; Sharpley et al. 1997). N leaching rates where animal manure or sewage sludge is applied based on crop N requirements are consistently higher than where crop N needs are met with inorganic N sources. In all the above scenarios, N leaching to groundwater occurs as a result of elevated concentrations of NO_3 in the soil profile during groundwater recharge. Therefore, strategies that minimize pore water NO_3 concentrations at the onset of groundwater recharge are keys to reducing rates of NO_3 losses from agricultural systems. For most regions of the US and Europe, evapotranspiration rates exceed average rates of precipitation during the growing season (May through September). Therefore, the primary factor dictating N leaching rates is the availability of NO_3 in the soil profile following fall crop harvest.

A multistate commitment for the reduction of eutrophication in the Chesapeake Bay made in 1987 called for a 40% reduction of controllable N and P inputs by the year 2000 (Boesch et al. 1999). Each state has implemented a variety of programs to reduce agricultural nutrient sources. Maryland's program consisted of a volunteer N-based nutrient plan that relied on matching N inputs to crop demand by fertilizing for realistic yields, splitting N applications and using a pre-sidedress nitrogen test (PSNT) to determine additional mid-season N requirements. As of 1999, N-based nutrient management plans had been written and implemented for approximately 400000 hectares. Unfortunately, N concentrations in groundwater, streams, and rivers in areas dominated by agricultural inputs have not decreased. Uncertainty in weather, droughts, disease, insect pressure, and fall mineralization of the organic pool following crop harvest limit the long-term potential for significant reductions in N losses based on matching N inputs with crop demands. Furthermore, research suggests that N losses to groundwater following soybean harvest (for which there is no N application) can be as great as following crops for which N is applied.

Reducing fertilizer application rates, optimum timing of fertilizer application, crop rotation, and the use of winter cereal cover crops (Figure 3A) could result in approximately a 20% reduction in the discharge of N to groundwater, streams and rivers (Staver and Brinsfield 1994; Mitch et al. 1999). If further reductions in N inputs to crops are required, crop yields would very likely be reduced below the economic optimum. N losses from fields increase substantially as the crop requirements are approached and exceeded. Consequently, agronomic prescriptions that closely match the amount and timing of crop requirements provide significant environmental benefits, but with modest cost savings in terms of reduced fertilizer use and increased costs of testing. It is often cheaper to avoid risks of lowered yields and somewhat overapply fertilizers. Precision agricultural practices (Schepers, this volume) offers benefits, in addition to fertilizer savings, of increased yields. In addition, in the United States there is active consideration of other steps that would reduce application of nutrients in excess of crop requirements, such as insurance coverage that would compensate farmers if adherence to a rigid prescription reduced crop yields.

In addition to reductions from on-farm practices, other techniques, including the creation and restoration of wetlands and riparian ecosystems between farmland and streams and rivers and downstream, offer real opportunities for additional significant N reductions from agriculture. For example, research by several investigators (Jørgensen 1994; Lowrance et al. 1997) has shown significant N reductions using combinations of grass and tree buffer zones between farm fields and streams. Although significant N reductions are possible, optimum position of the riparian buffer or wetland in the landscape is critical (Staver and Brinsfield 1996; Mitsch et al. 1999). Mitch et al. estimate that creating or restoring 21,000 to 52,000 km^2 of wetlands in the Mississippi River basin (0.7 to 1.8% of the basin) would result in a 20 to 50% reduction in N loading to the Gulf of Mexico. While ecosystem restoration strategies have tremendous potential, adequately compensated volunteer programs, including permanent easements, must be available to the landowner.

Similar struggles to balance food production and water quality are occurring in Europe. In Germany, where rising NO_3 concentrations in groundwater are the most immediate concern, use of commercial N fertilizer grew about four fold from 1951 to 1990, while the estimated N content of farm produce increased just over two fold (van der Ploeg et al. 1997). Thus, N-use efficiency decreased and the surplus available to contaminate goundwater and surface waters increased greatly. The European Union nations are required by the Union's 1991 Nitrate Directive to have a voluntary code of good agricultural practices (Council of European Communities 1991). They are also required to identify vulnerable zones where additional action must be taken to reduce N losses to acceptable levels. More recently, a team of six European research institutions began evaluating economic alternatives for reducing N losses from agriculture. The so-called Nitrotax study is coordinated by the Centre for Agriculture and Environment in The Netherlands. Researchers are using a combination of farm models, literature reviews and expert judgment to better understand the implications of economic systems for N control. One of the goals is to form a "European Community" which seeks to merge individual countrie's agricultural programs into a unified effort to develop fair N policies that stabilize farm incomes and simultaneously meet regional water quality goals.

Phosphorus

While N losses to the environment from agriculture are widespread in most developed countries, most P losses occur in areas where livestock and poultry are highly concentrated, sewage sludge is applied or where there are high rates of soil erosion. Unlike N, which is highly dependent on dynamic soil microbial processes, P levels in soils as well as in manure and sewage sludge are easier to quantify and track. Furthermore, soil P levels change slowly, allowing for development of mass balances within fields, farms, and watersheds.

In the context of ecosystem protection, strategies for P reductions have focused mainly on educational and incentive programs to encourage implementation of practices designed primarily for reducing the movement of sediment from cropland (reduced tillage, grassed waterways, retention ponds, terraces, and contours). This strategy has been successful for reducing sediment and sediment-bound P losses on cropland where erosion rates are high. However, research suggests that these strategies for soils with high P levels not predisposed to high erosion rates may be less effective.

Soil P levels are, in general, increasing throughout Europe and the U.S., particularly where livestock and poultry manure or sewage sludge is applied to cropland. Crop nutritional requirements and grain removal rates indicate a N:P supply ratio of approximately 6:1 on a weight basis, which is twice the ratio of plant-available N:P in manure or sewage sludge. Consequently, repeated applications of animal manure or sewage sludge to cropland, based on N crop requirements, results in unused P and increasing soil P levels. For example, in the states of Maryland and Delaware, many soils have soil P levels sufficient to meet crop P requirements for several years without additional P fertilization (Sims 1987, 1993). Problems associated with high soil P levels are exacerbated because they are often located near sensitive freshwater bodies in which P enrichment causes excessive algal growth.

On the Delmarva Peninsula, an intensive poultry producing region lying between the Atlantic Ocean and the Chesapeake Bay, repeated application of poultry litter has been increasing topsoil P inventories by an average of 10 kgha^{-1} annually beyond P removal rates required for the optimum production of crops (Kenneth Staver, pers comm). From the perspective of the total regional P budget, approximately one-half of the total P imported to the Maryland portion of the peninsula in the form of fertilizer, sewage sludge, grain, and dietary supplements for poultry remains as a residual in the soils. Similar studies in Denmark, Norway, Finland and Sweden suggests that the main reason for high P losses from cropland is the net input of approximately 20 kg-ha^{-1} annually for the past several decades (Svendsen and Kronvang 1991). Iserman (1990) reported even higher P surpluses for Germany and the Netherlands.

The potential for P loss from these soils is worsened by high water tables, tile drainage, and soils with low water-holding capacity. Breeuwsma et al. (1989) estimated that for a watershed where over 80% of the soils were P-saturated, leaching losses of 2.5 kg-ha^{-1} and runoff losses of 0.3 kg-ha^{-1} of total P were observed annually. For soil types not predisposed to high erosion rates, reductions in surface runoff of P will require better management of P levels in the uppermost soil horizon. However, one of the most effective best management practices for erosion control (no-till) actually increases P levels in the top soil horizons and under certain conditions can increase dissolved P losses. In a study by Staver and Brinsfield (1994), total P losses from fields under no-till for many years was twice as high on an annual basis compared to conventional till (Figure 3B). When toxic *Pfiesteria piscicida* outbreaks occurred in subestuaries of the Chesapeake Bay along the Delmarva Peninsula in 1997, the combination of highly P-enriched soils in the region, emerging evidence regarding loss of dissolved P from enriched soils, and high levels of P in the estuarine waters provided the impetus for legislation in Maryland mandating management of manure applications based both on N and P crop requirements.

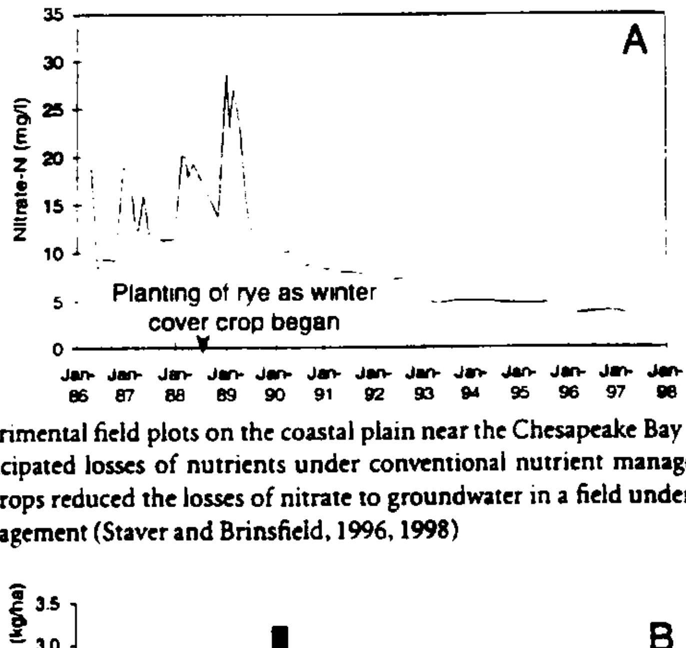

Fig. 3A: Experimental field plots on the coastal plain near the Chesapeake Bay have demonstrated unanticipated losses of nutrients under conventional nutrient management: A. Fall cereal cover crops reduced the losses of nitrate to groundwater in a field under conventional nutrient management (Staver and Brinsfield, 1996, 1998)

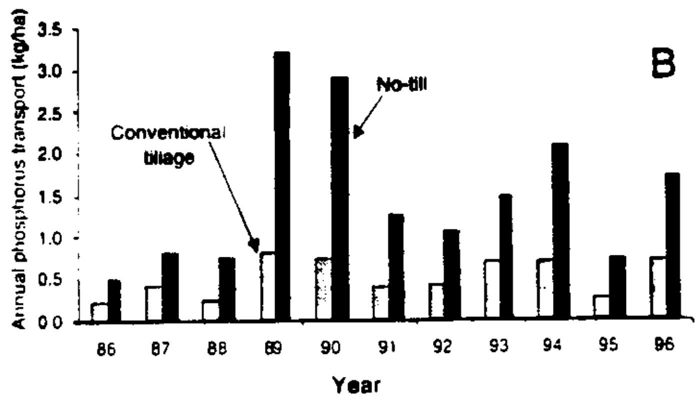

Fig. 3B: No-till field management resulted in increased phosphorus concentrations in surface soils, with increased losses of dissolved P in runoff compared to tilled fields (Staver and Brinsfield, 1994, updated)

While strategies which reduce sediment-bound P transport are important in the long term, better management of P inputs on a watershed basis will be required. This is especially true for managing manure in regions with concentrated animal production. A more holistic systems approach is needed. Efforts should include alternative uses for manure including burning, pelletizing, and composting and further research on feed, nutrition, and manure handling and storage. Nutrient composition of manure can be altered by changing diet composition to provide a better balance in nutrient output. Greater feed efficiency is generally economically desirable and can reduce the quantity of manure. Recent poultry diet research by Sims et al. (1999) has shown that the combination of the enzyme phytate and low phytic acid corn reduced total P in poultry manure by 50% and soluble P by 84% when compared to traditional diets. If the P in manure could be reduced by 50% below current levels, then the N:P ratio of the manure would be closely balanced with crop N:P requirements, thereby reducing the buildup of soil P following repeated applications, based on crop N requirements.

Integrated Strategies

Managing nutrients (both N and P) in farm fields, feedlots and the environment to achieve significant reductions in nutrient inputs from agriculture requires integrated strategies. The problems related to phosphorus accumulation in soils managed solely for nitrogen and the recognition of distant eutrophication driven by nitrogen losses from fields under P-based management demonstrate the need to manage both nutrients in an integrated manner. Furthermore, some nutrient losses from agricultural activities, whether they originate from fertilizers, mineralization of soil organic matter, or animal wastes are inevitable. More efficient agronomic practices, moderation of excessive soil draining, and improved animal waste handling and disposal could result in reductions of nutrient loadings to coastal waters of 20 to 30%. Also, nutrient traps or sinks could be restored and enhanced in watersheds under intense agricultural production in order to capture some of the nutrients inevitably lost downstream. Expanded restoration and creation of wetlands and riparian forests and reopening flood plains could trap another 20-30% of the nutrients. Thus, reduction of nutrient inputs from agricultural activities at the present level of intensity of 50% or more is potentially feasible. However, this will require substantial technical and financial assistance and greatly increased governmental and private efforts to restore and manage critical nutrient-trap habitats (Keeney and DeLuca 1993; Mitsch et al. 1999). In the process, the esthetics, wildlife, and fisheries production, and biodiversity of "overdeveloped" agricultural landscapes could be enhanced.

In an economic assessment of the costs of agricultural nutrient controls in the Mississippi River Basin, Doering et al. (1999) concluded that N losses could be reduced by 20% through a combination of fertilizer restrictions and wetland restoration with minimal net economic impact and no loss of production. Based on this model, reductions in production would only become significant when N loss restrictions reach 30% and higher, but the effects on aggregated producer net returns would be partially offset by increased commodity prices. In practice, though, it is the returns to individual producers rather than the aggregate that determine the economic viability of agriculture. The inequities are further compounded because it is more effective for reducing downstream nutrient loads in downstream nutrient loads to target nutrient use restrictions and transformation of agricultural lands to wetlands geographically rather than to apply these efforts uniformly throughout the catchment basin. These conditions challenge society to find ways to internalize the heretofore ignored external environmental costs within agricultural commodity prices and to provide targeted technical assistance, subsidies and incentives to achieve environmental benefits while mitigating inequities.

Large Watershed Management

The mounting pressures to control nutrient inputs to coastal waters, including those from inputs from agriculture, will only grow in the future. Although major gains in environmental quality have been realized by reducing organic and toxic loading from municipal and industrial waste discharge, advanced wastewater treatment to remove nutrients (especially N) has been accomplished in only a few regions. However, reduction of sources of P (elimination of phosphate-based detergents) and chemical and biological treatment to reduce both P and N are being shown to be feasible and successful. While significant reductions in N and P from wastewater treatment plants have occurred, the application of sludge generated by these processes may result in losses of these nutrients from croplands on which the sludge is applied. There is even promise that atmospheric inputs of N will be reduced due to air quality regulations requiring greater reduction of nitrous oxide emissions from both stationary sources and motor vehicles.

Major challenges still confront the reduction of agricultural nonpoint sources of nutrients. These sources are numerous and diffuse. Agriculture is economically and socially important but is undergoing serious conflicts between traditional operations and global economic forces. Consequently, production agriculture is generally marginally profitable, at best, and can ill afford increased costs for environmental controls. The market place does not easily allow the transfer of these external environmental costs to the consumers, who desire cheap food prices and have many choices. The impacts of agricultural nutrient pollution are often far removed (thousands of kilometers in some cases) from their sources, so that the usual forces of place-based concerns and community equity are not effective in motivating pollution abatement. The causes and effects transcend multiple political boundaries, across nation states and subnational governments – and so does the need for common solutions.

For all of these reasons, beginning in the late 1980s and the 1990s, abatement of large-scale coastal eutrophication has been addressed by a variety of ecosystem management efforts, including regional seas programs and national or watershed-scale assessments, management and regulation. Although eutrophication is often a major concern and a driving issue in these efforts, multiple issues are usually dealt with, including pollution by toxins, oil spills, fisheries management, habitat protection and restoration, and even social and livability issues.

The oldest of these multijurisdictional management endeavors focuses on the Baltic and North Seas. The North Sea Ministerial Conferences operating under the Paris Convention has set goals for a 50% reduction of contaminants discharged into the sea. The Baltic Marine Environmental Protection Commission, or Helsinki Commission (HELCOM) has nine nations plus the European Union as members. Although much of its effort has focused on activities in the Baltic itself and around its coasts, increasing attention is being paid to watersheds draining into the Baltic. HELCOM has issued recommendations for agricultural nutrient management that the member nations are expected to adopt. The Black Sea Environmental Programme has six littoral member nations, but the catchment area for the Black Sea includes some 17 nations and more than 160 million inhabitants. The Danube River Programme reaches upstream to involve 11 nations in improving the environmental quality of the river, its delta, and its effects on the nearshore Black Sea. In Japan, the five prefectures bordering the Seto Inland Sea have a similar multiparty compact for common commitments and shared resources.

In the United States, the National Estuary Program has engaged state and local governments and the public in the development of comprehensive conservation and management plans for 28 bays and estuaries around the coasts. Eutrophication is a major issue being addressed in most of them. The oldest and largest regional coastal ecosystem restoration is that of the Chesapeake Bay Program, which engages three states, the District of Columbia, and the federal government. Reversing eutrophication is the highest priority, with a commitment made in 1987 to reduce controllable sources of inputs of both nitrogen and phosphorus by 40% by 2000 (Boesch et al. 1999). A substantial element of the program seeks to provide technical assistance to farmers to reduce nutrient losses to the Bay. As the 2000 milestone approaches, models are being run and monitoring data analyzed to determine whether the goal would be met. It now appears that the phosphorus target will be met or exceeded and that – although N loadings have been reduced – progress will fall short of the goal for N (Boesch et al. 1999). Most documented gains have been made through reduction of point sources.

Even more aggressive commitments for nutrient reduction have been made in Denmark, where in 1987 the Parliament enacted the Action Plan on the Aquatic Environment, the main objectives of which were to reduce total nitrogen discharge to the aquatic environment by 50% and the total discharge of phosphorus by 80% by 1994 (Kronvang et al. 1993). While the goal for reductions of phosphorus, most of which were from point sources, has been met, it is estimated that the nitrogen load has been reduced by only about 15% (Conley and Josefson, 1999). These nitrogen reductions were mainly due to improved sewage treatment; little progress has been made on reductions of agricultural sources, which dominate the nitrogen loads. The second phase of the Action Plan was begun in 1998 and will include requirements for maximum livestock concentrations, manure storage facilities, mandatory fertilizer plans and budgets, cover crops, and set asides of agricultural lands.

Another large-scale ecosystem restoration focuses on the lower Florida peninsula and involves efforts to restore freshwater flows and improve water quality in the Everglades. Protection and restoration of Florida Bay, a shallow tropical ecosystem that has experienced discoloring algal blooms in recent years (Boesch 1996), is also an objective. Nutrient inputs from sugarcane production and other agricultural activities to the north are receiving particular attention and may be related to the problems seen in Florida Bay. Undoubtedly, the geographically largest challenge for the control of agricultural nutrient inputs concerns the eutrophication of the northern Gulf of Mexico. Ongoing assessments are evaluating the causes, consequences, sources, control options, and economics of agricultural nutrient reductions. It is clear that the majority of the nutrients discharged by the Mississippi and Atchafalaya rivers emanate from intensely farmed regions in the upper basin, over 1000 km upstream (Goolsby et al. 1999).

Some of these programs have demonstrated progress in arresting, if not reversing, coastal eutrophication. Most are just starting, and many face daunting challenges as a result of scale of the problems, fragmented responsibilities, and modest economic ability to address the needed controls. Most of the reductions in nutrient loading, even in the more advanced efforts such as the Chesapeake Bay Program, have come as a result of point source reductions. Pressures will intensify to redouble our efforts to control agricultural sources of nutrients – a major issue for agriculture during the first half of the 21st century.

References

Anderson DM (1995) Toxic red tides and harmful algal blooms: a practical challenge in coastal oceanography. Rev Geophysics 33:1189-1200

Angle JS (1990) Nitrate leaching losses from soybeans (*Glycine max* L. Merr.). Agric Ecosys Environ 31:91-97

Asman WAH, Larsen SE (1996) Atmospheric processes. In Jørgensen BB and Richardson K (eds.) Eutrophication in coastal marine ecosystems. American Geophysical Union, Washington, DC, p. 21-50

Baker JL, Johnson HP (1981) Nitrate-nitrogen in tile drainage as affected by fertilization. J Environ Qual 10:519-522

Balkas T, multiple authors (1990) State of the marine environment in the Black Sea region. UNEP Regional Seas Rep Studies 124:41 p

Boesch DF (1996) Science and management in four U.S. coastal ecosystems dominated by land-ocean interactions. J Coastal Conserv 2:103-114

Boesch DF, Brinsfield RB, Magnien RE (1999) Chesapeake Bay eutrophication: scientific understanding, ecosystem restoration, and challenges for agriculture. J Environ Qual, in press

Borum J (1996) Shallow waters and land/sea boundaries. In Jørgensen BB, Richardson K (eds) Eutrophication in coastal marine ecosystems. American Geophysical Union, Washington, DC, pp 179-203

Breeuwsma A, Reyerink JGA, Brus DJ, van heet Loo H, Schoumans OF (1989) Phosphate loading of soils, groundwater and surface waters in the catchment area of the Schuitenbeek (in Dutch). DLO Winand Staring Centre, Report 10, Wageningen, The Netherlands

Bricker SB, Clement CG, Pirhalla DE, Orlando SP, Farrow DRG (1999) National estuarine eutrophication assessment: a summary of condition, historical trend and future outlook. National Oceanic and Atmospheric Administration, Silver Spring, Maryland.

Burkholder JM, Glasgow HB, Jr (1997) *Pfiesteria piscicida* and other *Pfiesteria*-like dinoflagellates: behavior, impacts, and environmental controls. Limnol Oceanogr 42:1052-1075

Caddy JF (1993) Toward a comparative evaluation of human impacts on fishery ecosystems of enclosed and semi-enclosed seas. Rev Fish Sci 1:57-95

Cambardella CA, Moorman TB, Jaynes DB, Hatfield JL, Parkin TB, Simpkins WW, Karlen DL (1999) Water quality in Walnut Creek watershed: nitrate-nitrogen in soils, subsurface drainage water and shallow groundwater. J Environ Qual 28:25-34

Comin RA, Romero JA, Astorga V, Garcia C (1997) Nitrogen re[illegible]val and cycling in restored wetlands used as filters of nutrients for agricultu[illegible] runoff. Wat Sci Technol 35:255-261

Conley DJ, Josefson AB (1999) Hypoxia, nutrient management and restoration in Danish waters. In Rabalais NN, Turner RE (eds) Effects of hypoxia on living resources, with emphasis on the northern Gulf of Mexico. American Geophysical Union, Washington, DC, in press

Cooper SR (1995) Chesapeake Bay watershed historical land use: impact on water quality and diatom communities. Ecol Appli 5:703-723

Cooper SR, Brush GS (1991) Long-term history of Chesapeake Bay anoxia. Science 254:992-996

Costanza R (1992) Toward an operational definition of ecosystem health. In Costanza R, Norton BG, Haskell B (eds). Ecosystem health – New goals for environmental management. Island Press, Washington, DC, p 239-256

Council of the European Communities (1991) Council Directive 91/676/EEC concerning the protection of waters against pollution caused by nitrates from agricultural sources. Official Journal No. L 375; Finnish Special Edition (15) 10:192

Diaz RJ, Rosenberg R (1995) Marine benthic hypoxia: a review of its ecological effects and the behavioural responses of benthic macrofauna. Oceanogra Mar Biol Ann Rev 33:245-303

Doering OC, Diaz-Hermelo F, Howard C, Heimlich R, Hitzhusen F, Kamierczak R, Lee J, Libby L, Milon W, Prato T, Ribaudo M (1999) Evaluation of economic costs and benefits of methods for reducing nutrient loads to the Gulf of Mexico. In Hypoxia in the Gulf of Mexico. NOAA Coastal Ocean Program Decision Analysis Series No., National Oceanic and Atmospheric Administration, Silver Spring, Maryland. (www.nos.noaa.gov/Products/pubs_hypoxia.html)

Duarte CM (1995) Submerged aquatic vegetation in relation to different nutrient regimes. Ophelia 41:87-112

Fail JL, Jr, Haines BL, Todd RL (1987) Riparian forest communities and their role in nutrient conservation in an agricultural watershed. Am J Alternative Agric 2:114-121

GESAMP (1990) The state of the marine environment. UNEP Regional Seas Reports and Studies No. 115. United Nations Environmental Program, Paris

Goda T (1991) Management and status of Japanese public waters. Water Sci Technol 23:1-10

Goolsby DA, Battaglin WA, Lawrence GB, Artz RS, Aulenbach BT, Hooper RP (1999) Flux and sources of nutrients in the Mississippi-Atchafalaya River basin. In Hypoxia in the Gulf of Mexico. NOAA Coastal Ocean Program Decision Analysis Series No., National Oceanic and Atmospheric Administration, Silver Spring, Maryland. (www.nos.noaa.gov/Products/pubs_hypoxia.html)

Grattan LM, Oldach D, Perl TM, Lowitt MH, Matuszak DL, Dickson C, Parrott C, Shoemaker RC, Kaufman CL, Wasserman MP, Hebel JR, Charache P, Morris Jr JG (1998) Learning and memory difficulties after environmental exposure to waterways containing toxin-producing *Pfiesteria* or *Pfiesteria*-like dinoflagellates. Lancet 352:532-539

Groffman PM (1994) Denitrification in freshwater wetlands. Curr Top Wetland Biogeochem 1:15-35

Heip C (1995) Eutrophication and zoobenthos dynamics. Ophelia 41:113-136

Hill AR (1996) Nitrate removal in stream riparian zones. Journal of Environmental Quality 25:743-755

Houde ED, Rutherford ES (1993) Recent trends in estuarine fisheries: predictions of fish production and yield. Estuaries 16:161-176

Howarth RW, Billen G, Swaney D, Townsend A, Jaworski N, Lajtha K, Downing JA, Elmgren R, Caraco N, Jordan T, Berendse F, Freney J, Kudeyarov V, Murdoch P, Zhu Z-L (1996) Regional nitrogen budgets and riverine N & P fluxes for the draina-ges to the North Atlantic Ocean: natural and human influences. Biogeochemistry 35:75-139

Hunt PG, Stone KC, Humenik FJ, Matheny TA, Johnson MH (1999) In-stream wetland mitigation of nitrogen contamination in a USA coastal plain stream. J Environ Qual 28:249-256

Iserman K (1990) Share of agriculture in nitrogen and phosphorus emissions into surface waters of Western Europe against the background of their eutrophication. Fert Res 26: 253-269

Jansson B-O (1997) The Baltic Sea: current and future status and impact of agriculture. Ambio 26:424-431

Jansson AM, Andersson R, Berggren H, Leonardson L (1994) Wetlands and lakes as nitrogen traps. Ambio 23:20-325

Jansson B-O, Dahlberg K (1999) The environmental status of yhe Baltic Sea in the 1940's, today and in the future. Ambio 28:312-319

Jordan TE, Weller DE (1996) Human contributions to terrestrial nitrogen flux. BioScience 46:655-664

Jørgensen BB, Richardson K (eds) (1996) Eutrophication in coastal marine ecosystems. American Geophysical Union, Washington, DC

Jørgensen SE (1994) A general model for nitrogen removal by wetlands. In: Mitsch WJ (ed) Global wetlands: Old World and New. Elsevier, Amsterdam, p. 575-583

Justić D (1991) Hypoxic conditions in the northern Adriatic Sea: historical development and ecological significance. In Tyson RV, Pearson TH (eds) Modern and ancient continental shelf anoxia. Geological Society Special Publication 58, The Geological Society, London, pp 95-105

Justić D, Legovic T, Rottini-Sandrini L (1987) Trends in oxygen content 1911-1984 and occurrence of benthic mortality in the northern Adriatic Sea. Estuarine Coastal Shelf Sci 24:435-445

Justić D, Rabalais NN, Turner RE (1994) Riverborne nutrients, hypoxia and coastal ecosystem evolution: biological responses to long-term changes in nutrient loads carried by the Po and the Mississippi Rivers. In Dyer KR, Orth RH (eds) Changes in fluxes in estuaries: implications from science to management. Olsen & Olsen, Fredensborg, Denmark, pp 161-167

Keeney DR, DeLuca TH (1993) Des Moines River nitrate in relation to watershed agricultural practices: 1945 versus 1980s. J Environ Qual 22:267-272

Kronvang B, Ærtebjerg G, Grant R, Kristensen P, Hovmand M, Kirkegaard J (1993) Nationwide monitoring of nutrients and thei. ecological effects: state of the Danish aquatic environment. Ambio 22: 176-

Larsson UR, Elmgren R, Wulff F (1985) Eutrophication and the Baltic Sea: causes and consequences. Ambio 14:9-14

Lowrance RL, Altier JD, Newbold R, Schnobel R, Groffman PM, Denver JM, Correll DL, Gillian JW, Robinson JM, Brinsfield RB, Staver KW, Lucas W, Todd AH (1997) Water quality functions of riparian forest buffers in Chesapeake Bay watersheds. Environ Management 21:687-712

Malone TC, Malej A, Harding Jr LW, Smodlaka N, Turner RE (1999) Ecosystems at the land-sea margin: drainage basin to coastal sea. American Geophysical Union, Washington, DC. 381 p

Mee LD (1992) The Black Sea in crisis: the need for concerted international action. Ambio 21:278-286

Mitsch WJ, Day JW, Gilliam Jr JW, Groffman PM, Hey DL, Randall GW, Wang N (1999) Reducing nutrient loads, especially nitrate-nitrogen, to surface water, groundwater, and the Gulf of Mexico. In Hypoxia in the Gulf of Mexico. NOAA Coastal Ocean Program Decision Analysis Series No., National Oceanic and Atmospheric Administration, Silver Spring, Maryland. (www.nos.noaa.gov/Products/pubs_hypoxia.html)

Nakata H (1995) Production enhancement and its implications for the restoration of marine biodiversity in the coastal waters of Japan. FAO Fisheries Circular No. 889. Food and Agriculture Organization, Rome

National Research Council (1994) Priorities for coastal ecosystem science. National Academy Press, Washington, DC

Nixon SW (1995) Coastal marine eutrophication: a definition, social causes, and future concerns. Ophelia 41:199-219

Nixon SW, Oviatt CA, Frithsen J, Sullivan B (1986) Nutrients and the productivity of estuarine and coastal marine systems. J Limnol Soc S Africa 12:43-71

Paerl HW (1995) Coastal eutrophication in relation to atmospheric deposition: current perspectives. Ophelia 41:237-259

Parker CA, O'Reilly JE (1991) Oxygen depletion in Long Island Sound: a historical perspective. Estuaries 12:248-264

Pearson TH, Rosenberg R (1978) Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. Oceanogr Mari Biol Ann Rev 16:229-311

Pinckney JL, Paerl HW, Harrington MB, Howe KE (1998) Annual cycles of phytoplankton community-structure and bloom dynamics in the Neuse River Estuary, North Carolina. Marine Biology 131:371-381

Rabalais NN, Turner RE, Justic D, Dortch Q, Wiseman Jr WJ, Sen Gupta BK (1996) Nutrient changes in the Mississippi River and system responses on the adjacent continental shelf. Estuaries 19:386-407

Richardson K (1996) Carbon flow in the water column case study: the southern Kattegat. In Jørgensen BB, Richardson K (eds) Eutrophication in coastal marine ecosystems. American Geophysical Union, Washington, DC, pp 95-114

Richardson K, Heilmann JP (1995) Primary production in the Kattegat: past and present. Ophelia 41:317-328

Ryther JH, Dunstan WM (1971) Nitrogen, phosphorus, and eutrophication in the coastal marine environment. Science 171:1008-1013

Seagle SW, Pagnotta R, Cross FA (1999) The Chesapeake Bay and northern Adriatic Sea drainage basins: land cover and nutrient export. In Malone TC, Malej A, Harding LW, Smodlaka Jr N, Turner RE (eds) Ecosystems at the land-sea margin: drainage basin to coastal sea. American Geophysical Union, Washington, DC, p 7-28

Schepers J (1999) Precision farming. OECD Conference on Biological Resource Management, Paris. This volume

Sharpley A, Meisinger JJ, Breeuwsma A, Sims JT, Daniel TC, Schepers JS (1998) Impacts of animal manure management on ground and surface water quality. In Hatfield J, Steward BA (eds) Animal waste utilization: effective use of manure as a soil resource. Sleeping Bear Press

Shepared SA, McComb AJ, Bulthuis DA, Neverauskas V, Seffersen DA, West R (1989) Decline of seagrasses. In Larkum AWD, McComb AJ, Shepard SA (eds) Biology of seagrasses: a treatise on the biology of seagrasses with special reference to the Australian region. Elsevier, Amsterdam, p 346-393

Sims JT (1987) Agronomic evaluation of poultry manure as a nitrogen source for conventional and no-tillage corn. Agron J 79:563-570

Sims JT (1993) Environmental soil testing for phosphorus. J Prod Agric 6:501-507

Sims JT, Layahun MF, Malone GW, Saylor WW, Rabov V (1999) Effects of microbial phytase, high available phosphorus corn, and dietary phosphorus level on broiler production. 3. Effects on broiler litter composition. Southern Poultry Science Society

Smayda TJ, Shimizu Y (1993) Toxic phytoplankton blooms in the sea. Elsevier, Amsterdam

Smith VH (1998) Cultural eutrophication of inland, estuarine, and coastal waters. In Pace ML, Groffman PM (eds). Successes, limitations, and frontiers in ecosystem science. Springer Berlin, Heidelberg, New York

Staver KW, Brinsfield RB (1994) The effects of erosion control practices on phosphorus transport from Coastal Plain agricultural watersheds. Chesapeake Research Consortium (Publication No. 149), Solomons, Maryland

Staver KW, Brinsfield RB (1996) Seepage of groundwater nitrate from a riparian agroecosystem in the Wye River Estuary. Estuaries 19: 359-370

Staver KW, Brinsfield RB (1998) Use of cereal grain winter cover crops to reduce groundwater nitrate contamination in the Mid-Atlantic Coastal Plain. J Soil and Water Conserv 3:230-240

Svendsen LM, Kronvang B (1991) Phosphorus in the Nordic countries: methods of bioavailability, effects, and measures. National Environmental Research Institute, Roskilde, Denmark. 201 pp

Tolmazin (1985) Changing coastal oceanography of the Black Sea. I. Northwestern shelf. Prog Oceanogr 15:227-276

Turner RE, Rabalais NN (1991) [illegible]anges in Mississippi River water quality this century: implications for coast[illegible] [illegible]od webs. BioScience 41:140-148

Ulanowicz RE (1997) Ecology: the ascendency perspective. Columbia University Press, New York

van Bennekom AJ, Salomons W (1981) Pathways of nutrients and organic matter from land to oceans through rivers. In Martin L J-M, Burton JD, Eisma D (eds.) River inputs to ocean systems. Food and Agriculture Organization, Rome, p 33-46

van der Ploeg RR, Ringe H, Machulla G, Hermsmeyer D (1997) Postwar nitrogen use efficiency in West German agriculture and groundwater quality. J Environ Qual 26:1203-1212

Vitousek PM, Aber JD, Howarth RW, Likens GE, Matson PA, Schindler DW, Schlesinger WH, Tilman DG (1997) Human alteration of the global nitrogen cycle: sources and consequences. Ecol Appl 7:737-750