Chapter 9

Nutrient Management

Coordinating Lead Authors: Robert Howarth, Kilaparti Ramakrishna
Lead Authors: Euiso Choi, Ragnar Elmgren, Luiz Martinelli, Arisbe Mendoza, William Moomaw, Cheryl Palm, Rabindra Roy, Mary Scholes, Zhu Zhao-Liang
Review Editors: Jorge Etchevers, Holm Tiessen

Main Messages **297**

9.1 Introduction **297**

9.2 Background **298**

9.3 Responses to Insufficient Nutrients to Support Agriculture in Some Regions **301**

9.4 Responses for Management of Excess Nutrients **301**
9.4.1 Leaching and Runoff from Agricultural Fields
9.4.2 Animal Production and Concentrated Animal Feeding Operations
9.4.3 Fossil Fuel Sources
9.4.4 Urban and Suburban Sources
9.4.5 Wetlands as Nutrient Interceptors: Enhancing the Sinks

9.5 Analysis and Assessment of Selected Responses **304**
9.5.1 Watershed-based versus Nationally Uniform Approaches
9.5.2 Voluntary Policies for Reaching Goals
9.5.3 Mandatory Policies for Reaching Goals: Regulations
9.5.4 Mandatory Policies for Reaching Goals: Taxes, Fees, and Marketable Permits
9.5.5 Hybrid Approaches for Reducing Coastal Nitrogen Pollution

9.6 Lessons Learned and Synthesis **308**

REFERENCES **309**

FIGURES

9.1 Global Trends in the Creation of Reactive Nitrogen through Human Activity, 1850–2100

9.2 Estimated Total Inorganic Nitrogen Deposition, Wet and Dry, in 1860, Early 1990s, and Projected for 2050*

9.3 Estimated Flux of Nitrogen to Coastal Waters from the Entire United States in Rivers and from Sewage Treatment Plants, 1960–2000, with Projections to 2030

9.4 Trends in Synthetic Nitrogen Fertilizer Use and Release of NO_x to the Atmosphere from Fossil Fuel Combustion in China, 1949–99

9.5 Crop Production and Leaching of Nitrogen to Surface and Ground Waters as a Function of Inputs to Agricultural Fields

TABLES

9.1 Increase in Nitrogen Fluxes in Rivers to Coastal Oceans Due to Human Activities for Some Contrasting Regions

*This appears in Appendix A at the end of this volume.

Main Messages

Human activity has greatly increased the flux of nutrients through the landscape, roughly doubling the global flux of nitrogen and tripling the flux of phosphorus in the landscape over natural values. Agriculture is the major driver of change for both of these nutrient cycles, although other factors also contribute, such as creation of reactive nitrogen during fossil fuel combustion and the use of phosphorus as a surfactant in detergents, with ensuing effects on the environment and human well-being. Much of the change is recent, and half the synthetic nitrogen fertilizer ever used on Earth has been utilized since 1985. The extent of alteration of nutrient cycling is not uniform over the planet and varies greatly from region to region. In many parts of Europe, Asia, and North America, nitrogen deposition from the atmosphere and nitrogen fluxes in rivers have increased 10-fold or more. On the other hand, in some regions where population levels are low and where there has not been much agricultural activity, little change, if any, has occurred in nutrient fluxes in the landscape. Some parts of the world, including much of Africa, suffer from too little fertilizer availability (particularly phosphorus fertilizer) to support agriculture needs, a stark contrast to the nutrient surpluses that characterize the developed world and East and South Asia.

The consequences of excess nutrient flows are large and varied. The effect on human health, while poorly quantified, is also varied and potentially severe. With phosphorus, the primary concern is eutrophication (excess algal growth) in freshwater ecosystems, which can lead to degraded habitat for fish and decreased quality of water for consumption by humans and livestock. For nitrogen, the range of issues is far greater. Ecological and environmental effects include eutrophication of coastal marine ecosystems, eutrophication of freshwater lakes in the tropics, contribution to acid rain with effects on both freshwater and terrestrial ecosystems, loss of biodiversity in both aquatic and terrestrial ecosystems, creation of ground-level ozone (which leads to loss of agricultural and forest productivity), destruction of ozone in the stratosphere (which leads to depletion of the ozone layer and increased UV-B radiation on Earth), and contribution to global warming. The resulting health effects include the consequences of ozone pollution on asthma and respiratory function, increased allergies and asthma due to increased pollen production, risk of blue-baby syndrome, increased risk of cancer and other chronic diseases from nitrate in drinking water, and increased risk of a variety of pulmonary and cardiac diseases from production of fine particles in the atmosphere.

Manifestations of these problems vary regionally, from too much exposure to nitrogen in the soils, atmosphere, and waters in much of industrial Europe and North America to nutrient shortages hurting subsistence farmers of Africa. Both extremes are found in Latin America and Asia, with the largest growth in demand and use of commercial fertilizers in Asia.

Technical tools exist for reduction of nutrient pollution at reasonable cost. That many of these tools have not yet been implemented on a significant scale suggests that new policy approaches are needed. Current regulatory authority for non-point source pollution is often nonexistent or very limited. Hence, increased authority to regulate such sources may be necessary to reverse pollution of surface water by nutrients. Reversal of soil nutrient depletion and consequent reduction of crop yield in Africa and many parts of Asia and Latin America can be realized through a combination of technology options and policy and institutional reforms.

Market-based instruments hold the potential for better nutrient management, but may not be relevant in all countries and circumstances. Relatively little is known empirically about the impact of these instruments on technological change. Also, much more empirical research is needed on how the pre-existing regulatory environment affects performance, including costs. Which instrument is best in any given situation depends upon the characteristics of the environmental problem, and the social, political, and economic context in which it is being regulated.

Policy responses to nutrient pollution can be addressed through uniform national approaches, or through a watershed-based approach. The latter is likely to be the most cost-effective for some sources of nutrients (such as runoff from agricultural yields), while a uniform national approach may be better for others (such as NO_x from fossil fuel combustion or phosphorus from detergents). An important question for the choice of the correct pollution control instrument is related to the implementation costs of the instruments. The more suitable measures are associated with implementation difficulties, and policy-makers might evaluate the trade-offs between cost-efficiency and ease of implementation. Policies should be developed to increase the supply of fertilizer to the regions where availability has been limited, and to encourage the fertilizer to be used efficiently and with less environmental leakage than has occurred in much of the industrial world. In industrial countries, policies need to be implemented to reduce this nutrient leakage. A major focus should be the increasingly concentrated production of animal protein in many regions, both since this can be a major driver in the increased use of synthetic nitrogen fertilizer and because the animal wastes are usually poorly treated and leak substantial amounts of both nitrogen and phosphorus to the environment.

Prospects need to be explored for developing a comprehensive understanding of a nutrient management strategy that transcends geographical, economic, and political boundaries to minimize the need for extracting phosphate from limiting reserves and for introducing more biologically available nitrogen into the biosphere and to distribute those nutrients efficiently according to local, regional, and global demands. Together with the understanding of a management strategy, the existing data on nutrient mobilization, distribution, and effects need to be assessed to insure that the science used to develop management strategies is sound and complete.

9.1 Introduction

Globally, the world has seen a tremendous increase in the use of synthetic nitrogen (N) fertilizer and inorganic phosphorus (P) fertilizer over the past half century. In conjunction with the inadvertent creation of reactive N during the combustion of fossils fuels, human activity has increased N fluxes two-fold and P fluxes three-fold (Vitousek et al. 1997; NRC 2000; Smil 2001; Howarth et al. 2002b; Galloway et al. 2004). However, these changes are far from uniform, and some regions have seen increased nutrient fluxes of ten-fold or more, while other regions have seen little or no increase (Howarth et al. 1996, 2002b; NRC 2000). Many regions of the world—most notably sub-Saharan Africa—have insufficient inputs of new nutrients to support agriculture. When crops are harvested, N and P are removed with the harvest, and insufficient return of these on nutrient-poor soils leads not only to low crop production but also to further degradation of soil quality due to increased erosion. Many other regions of the world—including most of the industrial world as well as East Asia, South Asia, and southeastern South America—now suffer from greatly increased fluxes of nutrients in aquatic ecosystems and, in the case of N, the atmosphere. (See MA *Current State and Trends,* Chapter 13.) It is important to note that while the world is divided into two parts—regions where nutrients flow in excess and regions where nutrient inputs are insufficient to provide adequate food production—this division does not simply follow the classical division between developed and developing nations. While the regions that have insufficient nutrient supplies fall within the developing world, many other developing regions have problems with surplus nutrients that match those of the industrial countries.

The consequences of excess nutrient flows are large and varied. For P, the primary concern is with eutrophication (excess algal growth) in freshwater ecosystems, which can lead to degraded habitat for fish and decreased quality of water for consumption by humans and livestock (Carpenter et al. 1998). For N, the range of issues is far greater and, in fact, a single atom of N can cascade through the environment and cause multiple problems (Galloway et al. 2003). Ecological and environmental effects include eutrophication of coastal marine ecosystems, eutrophication of freshwater lakes in the tropics, contribution to acid rain with effects on both freshwater and terrestrial ecosystems, loss of biodiversity in both aquatic and terrestrial ecosystems, creation of ground-level ozone (which leads to loss of agricultural and forest productivity), destruction of ozone in the stratosphere (which leads to holes in the ozone layer and increased UV-B radiation on Earth), and contribution to global warming (Vitousek et al. 1997; NRC 2000; Howarth et al. 2000; Tartowski and Howarth 2000; Rabalais 2002; Galloway et al. 2003).

The human health effects, while poorly quantified, are also varied and potentially severe. These include the consequences of ozone pollution on respiratory function, increased allergies and asthma due to increased pollen production, risk of blue-baby syndrome if nitrate levels in drinking water are high, increased risk of cancer and other chronic diseases from the presence of nitrate in drinking water, and increased risk of a variety of pulmonary and cardiac diseases from the production of fine particles in the atmosphere (Wolfe and Patz 2002, Townsend et al. 2003).

The problems from nutrient pollution on surface water quality are particularly well recognized, especially in industrial countries (Vitousek et al. 1997; Carpenter et al. 1998; Howarth et al. 2002a). The importance of P in promoting excessive aquatic plant production in fresh water was realized early, and was already well understood in the 1960s (Vollenweider 1968). Despite some early important studies (Ryther and Dunstan 1971), public awareness of the problems caused by excessive N inputs, however, took much longer to develop (Howarth and Marino, in press). While significant progress in reducing P pollution has been made in developed countries over the past 30 years, N pollution has actually grown worse, and is particularly evident in coastal ecosystems (NRC 1993a, 2000; Nixon 1995; Howarth et al. 2000). Today, two thirds of the coastal rivers and bays in the United States are believed to be moderately to severely degraded from excess N inputs (NRC 2000). Thus improving water quality is a major focus of nutrient management in industrial countries. .

In the developed world, P pollution has been counteracted increasingly effectively since the 1960s, through P precipitation in sewage treatment, bans on P in commercial detergents, and better controls on erosion of P-rich soils from agroecosystems. Some problem areas remain due to either excessive sewage inputs or inadequate control on soil erosion. Thus while many lakes have improved greatly, in some cases increasing agricultural P losses have resulted in continued deterioration (Foy et al. 2003). In contrast, N pollution is more of a problem with coastal marine ecosystems, a problem that many managers have only recently recognized (NRC 2000; Howarth and Marino, in press.). Thus there has been less effort and little success in stemming its rise. Nitrogen fluxes to surface waters are more difficult to control than P, due to their greater mobility through the atmosphere and groundwater (NRC 2000; Howarth et al. 2002b).

Worldwide, the most dramatic reversal of N pollution from non-point sources has been inadvertent. The end of the former Soviet Union led to the economic collapse of agriculture in Eastern Europe, and fertilizer use plummeted. As a result, nutrient loading to the Black Sea, especially of N, was greatly reduced. In only a few years, the Black Sea began to recover, including fish stocks and fisheries, from the eutrophication that had grown steadily worse from 1960 until 1990 (Boesch 2002). In developing countries, nutrient pollution has often gone unchecked, being low on the list of national priorities unless drinking water supplies are affected.

Much of the developing world did not take to scientific agriculture until the 1960s. With the expansion of education and training, India and China, in particular, have shown dramatic improvements in their agricultural practices with attendant increases in the use of commercial fertilizer. The introduction of improved varieties of wheat and rice that were responsive to fertilization (Hayami and Otsuka 1994), and the knowledge of successful application breakthroughs in the industrialized world involving fertilizers (Critchfield 1982), further contributed to the increased use of synthetic fixed N. Between 1960 and 1980, global production of wheat and feed grains grew slightly faster than the population, yielding a net increase in food supplies per person of 0.8% per year. China, in just three years in the late 1970s saw 15% growth in corn production, a 20% expansion in rice production, and a 40% gain in wheat production (Insel 1985).

Other parts of the developing world did not benefit to the same extent, with Africa providing many tragic examples of lost opportunities (Paarlberg 1996). Large areas of sub-Saharan Africa are affected by nutrient depletion (Stoorvogel and Smaling 1990). The consequences include low crop yields (Sanchez 2002) and increased erosion as a result of decreasing vegetative cover. Nutrient depletion has arisen through continual nutrient removal via crop harvests with insufficient nutrient replacement. This soil nutrient mining is in part due to the removal of fertilizer subsidies in the region in the 1990s through structural adjustment programs (Scoones and Toulmin 1999). A comparison of fertilizer use in 1999, surprisingly, shows a higher average application rate per hectare in developing countries as a whole (91 kilograms per hectare per year) than in industrial countries (87 kilograms) (Dudal 2002). However, sub-Saharan Africa is distinguished by an average rate of 9 kilograms per hectare per year, with a high of 56 for Zimbabwe and a low of 0.8 for Rwanda.

9.2 Background

Human activity has greatly accelerated the cycling of both N and P globally since the start of the industrial and agricultural revolutions. For P, the largest human influence has been increased erosion due to agriculture and the application of P fertilizer to agricultural lands, although the use of P as a surfactant in detergents also had a major influence, particularly before many governments banned them. Historical trends in increased global mobilization of P can be evaluated from oceanic sediment records. Globally, human activity has probably increased fluxes of P from land to the oceans three-fold, from a natural flux of 8 Tg P per year to the current flux of 22 Tg P per year (Howarth et al. 1995, 2002b; NRC 2000). The increase in the flux that is attributable to human activity (14 Tg P per year) is roughly equivalent to the rate at which P fertilizer is now used globally in agriculture (NRC 2000).

Human activity has also had an immense effect on the global cycling of N. This is mainly due to the use of synthetic fertilizer in agriculture. However, the release of N pollution during fossil-fuel combustion also contributes significantly (Vitousek et al. 1997; Galloway et al. 2004). Prior to the industrial revolution, most reactive N on Earth was created through the natural process of bacterial N fixation, with perhaps half occurring on land and half in the oceans (Vitousek et al. 1997; Cleveland et al. 1999; Karl et al. 2002; Galloway et al. 2004). Human activity has now roughly doubled the rate of creation of reactive N on the land surfaces of Earth. (See Figure 9.1.) The rate of change is extraordinarily rapid, and the N cycle is changing faster than that of any other element (Vitousek et al. 1997). More than half of all the synthetic N fertilizer ever used on the planet has been utilized since 1985 (NRC 2000; Howarth et al. 2002).

Human alteration of nutrient cycles is not uniform across the world. The greatest changes occur where human densities and human activities such as agriculture and fossil-fuel combustion are the greatest. Some of this variation can be clearly seen in the rates of N deposition, which are far higher in Europe, East and South Asia, eastern North America, and southeastern South America than elsewhere in the world. (See Figure 9.2 in Appendix A.) Human activity (fossil-fuel combustion and volatilization of ammonia to the atmosphere from farm-animal wastes) has increased the rate of deposition over natural rates 10-fold or more in many of these regions (Holland et al. 1999). The N flux in rivers provides one of the best-integrated measures of human influences on the N cycle (Howarth et al. 1996, 2002b; NRC 2000). Whereas human activity has resulted in essentially no change in the flux of N in the rivers of some regions, such as Labrador and Hudson's Bay in Canada, it has increased fluxes by five- to ten-fold or more in many regions of North America and Europe since the start of the industrial and agricultural revolutions. (See Table 9.1.)

Taking the United States as an industrial-country example, Figure 9.3 describes how the problem grew, stabilized, and started to grow again. Growth in population, intensification of agricultural activities, and increased emissions of oxidized N pollutants (NO_x) to the atmosphere from fossil-fuel combustion were major drivers behind the increases in fluxes of N to coastal ecosystems during the 1960s and 1970s (Howarth et al. 2002a). These drivers were relatively constant in the 1980s, particularly for NO_x emissions (EPA 2000), even though emissions of NO_x in the United States did not decline as much as other air pollutants after the passage of the Clean Air Act Amendments in 1970 (NRC 2000). About half of the NO_x emissions came from mobile sources, including automobiles, buses, trucks, and off-road vehicles, and 42% from electric power generation (EPA 2000).

In the late 1990s, the United States produced approximately one third of all the NO_x released from fossil-fuel combustion globally (Howarth et al. 2002a). Since the late 1980s, the flux of N from the rivers to the coasts has increased again in the United

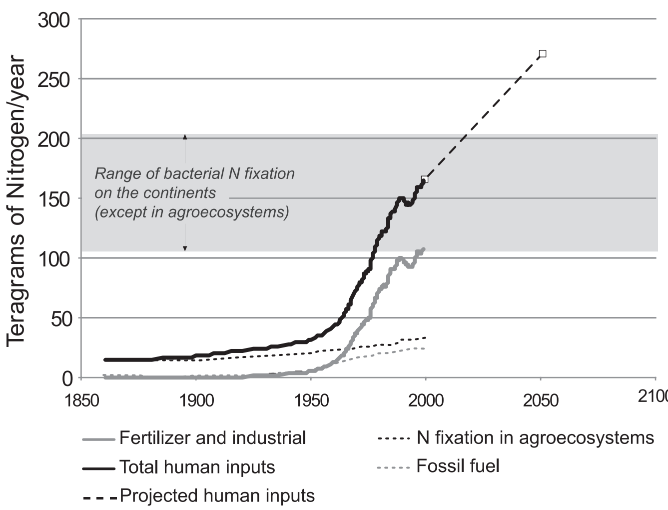

Figure 9.1. Global Trends in the Creation of Reactive Nitrogen through Human Activity, 1850–2100. The manufacturing of nitrogen by the Born Haber process for synthetic fertilizer and industrial use dominates, but the creation of reactive nitrogen as an inadvertent product of fossil-fuel combustion and the managed nitrogen fixation in agroecosystems also contribute. The natural rate of bacterial nitrogen fixation in natural terrestrial ecosystems (excluding fixation in agroecosystems) is shown for comparison. Note that human activity has roughly doubled the rate of formation of reactive nitrogen on the land surface of the planet. (Data for human creation of reactive nitrogen are from Galloway et al. 2004; data on natural rates of nitrogen fixation are from Vitousek et al. 1997 and Cleveland et al. 1999)

Table 9.1. Increase in Nitrogen Fluxes in Rivers to Coastal Oceans Due to Human Activities for Some Contrasting Regions (Howarth et al. 1996, 2002b; Bashkin 2002)

Region	Change
Labrador and Hudson Bay	no change
Southwestern Europe	3.7-fold
Great Lakes/St. Lawrence basin	4.1-fold
Baltic Sea watersheds	5.0-fold
Mississippi River basin	5.7-fold
Yellow River basin	10-fold
Northeastern United States	11-fold
North Sea watersheds	15-fold
Republic of Korea	17-fold

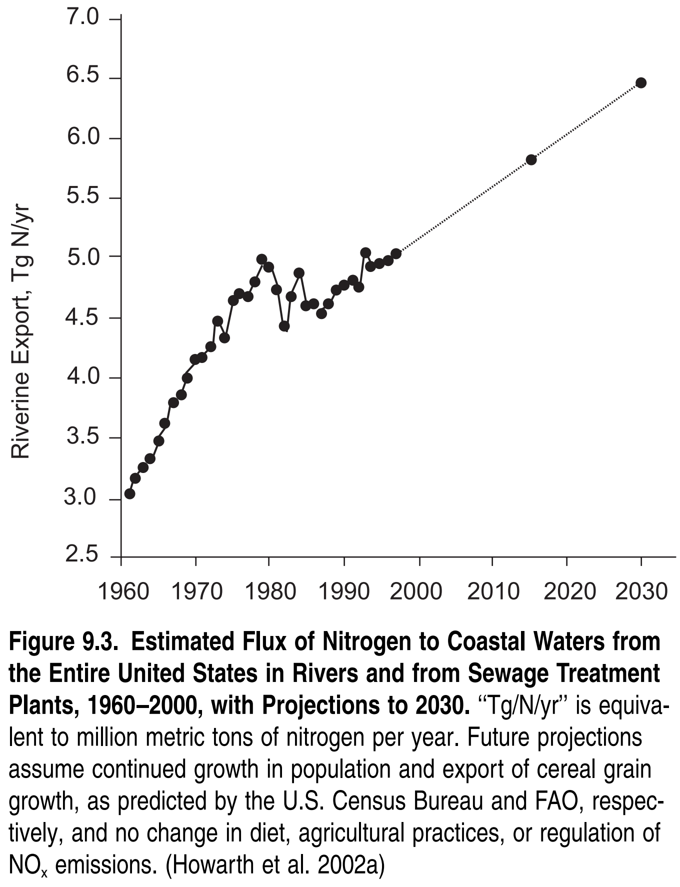

Figure 9.3. Estimated Flux of Nitrogen to Coastal Waters from the Entire United States in Rivers and from Sewage Treatment Plants, 1960–2000, with Projections to 2030. "Tg/N/yr" is equivalent to million metric tons of nitrogen per year. Future projections assume continued growth in population and export of cereal grain growth, as predicted by the U.S. Census Bureau and FAO, respectively, and no change in diet, agricultural practices, or regulation of NO_x emissions. (Howarth et al. 2002a)

States, largely due to increased synthetic fertilizer use and the increasingly industrialized production of meat protein (NRC 2000). Historical data for nitrate in major rivers such as the Mississippi River, the Susquehanna River (largest tributary of the Chesapeake Bay), and the Connecticut River (largest river input to Long Island Sound) show trends similar to those illustrated for total N use in the entire United States (Goolsby et al. 1999; Goolsby and Battaglin 2001; McIsaac et al. 2001; Jaworski et al 1997). If current trends in population growth, agricultural practices, grain exports, diet, and NO_x emissions continue, the flux of N to the coast is likely to continue to grow at the same rate as it has over the past decade (Howarth et al. 2002a). By 2030, N inputs to coastal waters in the United States could be 30% above the present level and over twice what they were in 1960.

The development of N pollution in Europe has followed a trajectory similar to that in the United States (von Egmond et al. 2002). It has been particularly troublesome for the shallow parts of the semi-enclosed seas, such as the Baltic Sea, the Black Sea, and the Adriatic Sea, whereas many coastal areas with strong tides and a good exchange of water with the open Atlantic have been much less affected. However, even the relatively open North Sea has experienced serious problems from eutrophication due to excess nutrient inputs.

Some developing countries have also experienced a rising N pollution problem as agriculture has intensified its use of synthetic fertilizers. The nitrate reaching the estuary of China's Changjiang (Yangtze) River increased four-fold from 1962 to 1990 (Shuiwang et al. 2000), and in 1995 an estimated 12 Tg of N was stored in agricultural soils in the major watersheds of China (Xing and Zhu 2002). Given current trends in N use and mobilization in China (Figure 9.4), these fluxes to the coast can be expected to continue to increase rapidly.

To date, there has been little progress in reversing the problem of coastal N pollution both in the United States (NRC 2000) and globally. N-removal technology for sewage treatment has led to

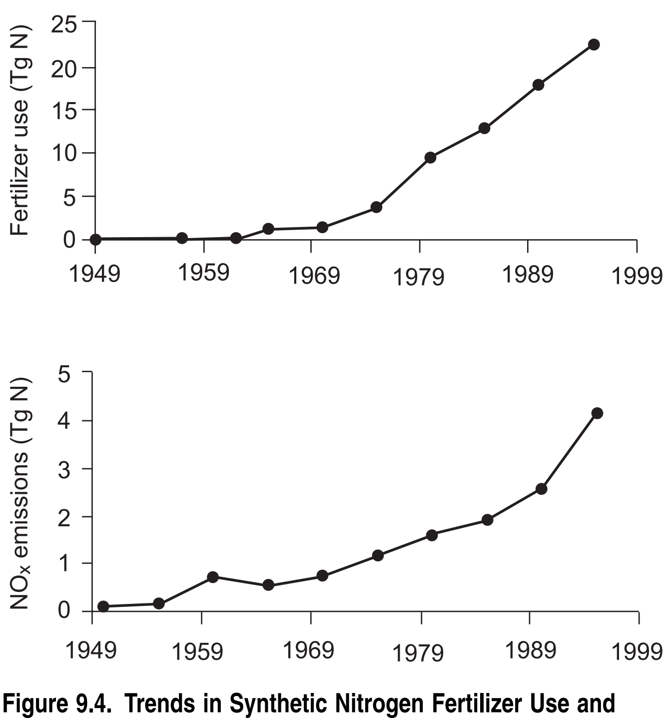

Figure 9.4. Trends in Synthetic Nitrogen Fertilizer Use and Release of NO_x to the Atmosphere from Fossil Fuel Combustion in China, 1949–99 (Xing and Zhu 2002)

water quality improvement in Tampa Bay and, to a lesser extent, in Chesapeake Bay in the United States (NRC 2000), and in Himmerfjärden Bay on Sweden's Baltic coast (Elmgren and Larsson 2001). Non-point sources of N, however, dominate inputs to most coastal waters of the United States and Europe (NRC 2000; Howarth et al. 1996, 2002b) and reducing them has proven problematic (Boesch et al. 2001a). In 1987, a target was set to reduce controllable inputs of N to Chesapeake Bay by 40% by the year 2000. This goal was not met, in part because management strategies for non-point sources were less effective than had been assumed (Boesch et al. 2001b) and because international agreements on 50% reductions for the North Sea (1987) and the Baltic Sea (1990) areas have also failed to reach their targets (OSPAR Commission 2000; Elmgren and Larsson 2001). One possible reason for these failures may be the long time needed for decreased nutrient losses from agriculture to show up as reduced riverine transport in some watersheds (Grimwall et al. 2000). Another reason may be an inadequate assessment of which management practices will work most effectively for N control (NRC 2000).

Many technical solutions for reducing N pollution exist, but few have been implemented systematically. In part, this is because nutrient management in both the United States and Europe has tended to concentrate on P pollution, which is the larger problem in freshwater ecosystems. The management community has been slow to recognize that N is often the larger problem in coastal marine ecosystems (NRC 2000; Howarth et al. 2000; Howarth and Marino, in press). Nitrogen is more mobile than P in the environment, flowing readily both through groundwater and the atmosphere, a difference in biogeochemistry that requires different management practices (NRC 2000).

Management of N pollution in coastal waters faces many policy challenges. Coastal ecosystems vary in their sensitivity to nutrient pollution, because of differences in the size of the watershed, in physical mixing regime, and in ecological structure (NRC 2000). Thus the same rate of N input will cause more harm in some locations than in others (Bricker et al. 1999; NRC 2000). Further, there is regional and local variation in the relative importance of N sources. Agriculture dominates input to the Mississippi River and is the primary cause of the hypoxic zone in the Gulf of Mexico (Goolsby et al. 1999; NRC 2000). Agriculture also dominates the inputs of N to the Baltic Sea and the North Sea, although human wastewater inputs are significant (Howarth et al. 1996). In many regions, including the Netherlands, Denmark, and Brittany in Europe and parts of the southeastern United States, animal wastes are a major part of these agricultural fluxes.

Atmospheric deposition of N from fossil-fuel combustion is the largest input to several marine ecosystems in the northeastern United States, mainly as runoff after deposition on the terrestrial landscape although, again, human wastewater inputs are important (Howarth et al. 1996; Jaworski et al. 1997; Boyer et al. 2002). And wastewater from sewage is the biggest input to the coastal waters of Korea (Bashkin et al. 2002) as well as to some major estuaries in the United States, including the Hudson River estuary in New York City and Long Island Sound (NRC 2000; Howarth et al. 2004) and rivers in the industrialized portion of Brazil (Martinelli et al. 1999; Martinelli 2003). Overall, human wastewater flows are thought to contribute 12% of the riverine N flux in the United States, 25% in Western Europe, 33% in China, and 68% in the Republic of Korea (Howarth et al. 1996, 2002a; NRC 2000; Bashkin et al. 2002; Xing and Zhu 2002). For many coastal rivers and bays, the relative importance of various inputs of N is uncertain, because the models used to estimate inputs are inexact and largely unverified (NRC 2000).

9.3 Responses to Insufficient Nutrients to Support Agriculture in Some Regions

Many regions of the world, particularly Africa, are in urgent need of greater nutrient inputs to support food production. The proper use of these increased nutrients would not only increase the regional food supplies but would also improve soil characteristics and, therefore, lead to less soil loss from erosion. The challenge is how to ensure that nutrient replenishment in developing countries does not follow the pattern of excessive nutrient applications that now threatens many ecosystems. In general, there is a need for balanced fertilizer applications tailored to specific soil nutrient deficiencies and agroecosystem requirements. Whether these nutrients are applied as mineral or organic fertilizers is not as important as the management practice that assures nutrient use efficiency both at the crop and the system level. Several nutrient replenishment strategies have been or are being developed in sub-Saharan Africa. The best strategy for nutrient replenishments will depend on the soil, climate, agroecosystem, socioeconomic conditions, and policy environment. Most of these nutrient replenishment strategies entail a combination of mineral and organic inputs, with the exact mix determined in part by socioeconomic conditions as well as the realization that organic materials cannot, in general, supply sufficient P to meet crop demand (Palm et al. 1997).

Though N is the primary limiting nutrient in most soils of sub-Saharan Africa, once the N is replenished, P quickly becomes limiting to crop production (Bationo et al. 1986). P can be the primary limiting nutrient in some of the sandy soils of the semi-arid area and on moderate-to-high P-fixing soils in the subhumid and humid areas (Buresh et al. 1997; Sahrawat et al. 2001). Several strategies exist for replenishing soil P: application of soluble P fertilizers, application of reactive phosphate rock (RP), or the combination of soluble P fertilizer and RP (Buresh et al. 1997). The replenishment can be achieved through a single large application with residual effects lasting several seasons or through smaller seasonal applications.

The direct application of phosphate rock is often proposed as the better alternative because of lower production costs than for soluble P fertilizers (Buresh et al. 1997). Phosphate rock deposits are found throughout Africa but they vary in their effectiveness for direct application to the soil (Mokwunye and Bationo 2002). The agronomic efficiency of the phosphate rock depends on its mineralogy, particle size, and reactivity (sedimentary forms being more reactive than igneous) as well as the soil type. Dissolution of the phosphate rock requires soils that are slightly acidic and low in calcium (Ca) and P in solution. The reactivity of the RP can also be increased through partial acidulation (Buresh et al. 1997) and also when used in combination with plants, particularly legumes, that are more efficient in accessing P (Lyasse et al. 2002).

The choice of P fertilizer depends, then, on the soil, the climate, plant species, and the comparative costs. While organic inputs do not have sufficiently high concentrations of P to replenish soil P at reasonable application rates, they can increase soil P availability above that obtained through the same application rates of mineral P. Where it is difficult and/or uneconomic to obtain P fertilizers, the combined use of organic and mineral P fertilizers has been shown to have higher P use efficiencies (Palm et al. 1997).

High rates of P application are likely to have negative environmental effects, primarily through erosion and runoff. Introduction of biological filter strips or biological terraces have proven quite effective in practically eliminating runoff and soil erosion of P; in addition, application of P increases the vegetative cover, practically eliminating runoff and loss of P by erosion. Losses of P through leaching are of concern primarily on sandy soils and medium textured soils with low soil organic matter; there is limited P movement down the soil profile on clay soils.

Biological N fixation offers an economically attractive alternative to synthetic N fertilizers (Bohlool et al. 1992; Döbereiner et al. 1995). Intercropping and rotation cropping is commonly done with N-fixing legumes. Nitrogen-fixing bacteria can be introduced in the soil to enhance N availability to both leguminous and non-leguminous plants. In Cuba, large-scale production and use of Azotobacter (free-living, N-fixing bacteria) is estimated to supply more than half of the N needed by non-legumes (Oppenheim 2001). Brazil has become the world leader in replacing chemical fertilizers with biological N fixation; mean value of N application is as low as 10 kilogram per hectare. Agriculture in Brazil is one of the main export activities, with soybeans, the largest export product of the country (Döbereiner 1997).

9.4 Responses for Management of Excess Nutrients

This section summarizes technical solutions for nutrient control and then outlines policy options for implementing them. The section relies heavily on the analysis by the U.S. National Research Council's Committee on Causes and Management of Coastal Eutrophication (NRC 2000) but uses other recent information in addition.

9.4.1 Leaching and Runoff from Agricultural Fields

Global use of synthetic N fertilizer increased steadily from about 11 teragrams N per year (million metric tons N per year) in 1960–61 to about 80 teragrams N per year in 1988–89, while fertilizer P use increased less steeply, from almost 5 to over 16 teragrams. Much of the recent growth has occurred in the developing world, particularly in China. The economic crisis after 1989 in the former Warsaw Pact countries led to an initial fall in global fertilizer use of 10–20%, but by 2000–01, global N use had recovered fully, and P use partly (IFA 2002). Global use of synthetic N fertilizer from 2001 exceeds 100 teragrams per year. Much of this fertilizer is assimilated by plants and harvested with crops, but some of it is lost to the environment. In the United States, for example, over half of the synthetic N fertilizer inputs are removed with crop harvest on average, and about 20% leaches to surface or ground waters (NRC 1993b; Howarth et al. 1996, 2002a).

The variability in leaching among fields is great, ranging from a low of 3% for grasslands with clay-loam soils to 80% for some row-crop agricultural fields on sandy soils (Howarth et al. 1996). P losses, while generally lower, are almost as variable (NRC 2000). Climate is important, with greater nutrient losses in areas of high rainfall and in wet years (Randall and Mulla 2001). These differences indicate that targeting particularly leaky types of agricultural fields can greatly reduce pollution of aquatic ecosystems.

Management practices for reducing N loss to downstream ecosystems have been reviewed by the National Research Council of the United States (NRC 1993b, 2000) and by Mitsch et al. (1999). The best management practices for reducing N pollution often differ from those for P, due to the higher mobility of nitrate-N in ground waters (NRC 2000). For instance, no-till agriculture reduces erosion and, therefore, P losses from fields, except at very high P inputs, but has little or no effect on nitrate loss (Randall and Mulla 2001). Particularly promising approaches for reducing N leaching from agricultural fields include:

- *Growing perennial crops such as alfalfa or grasses rather than annuals such as corn and soybeans.* Perennials retain N in the rooting zone and greatly reduce losses to groundwater. In Minnesota and Iowa, fields planted with perennial alfalfa lost 30- to 50-fold less nitrate than fields planted with corn and soybeans (Randall et al. 1997; Randall and Mulla 2001).
- *Planting winter cover crops, which greatly reduce the leaching of nitrate into groundwater during winter and spring, when most leaching normally occurs.* In a Maryland study, winter cover-crop plantings reduced nitrate loss three -fold (Staver and Brinsfield 1998).
- *Applying N fertilizer at the time of crop need.* In the North Temperate Zone, this is in the spring and summer, yet fertilizer, which is relatively inexpensive, is often applied in the fall, when the farmer has time, even if much of the applied fertilizer is leached to groundwater before crop growth begins in the spring. A study in Minnesota showed that fall application of fertilizer increased N leaching by 30–40% (Randall and Mulla 2001).

Much can be gained simply by eliminating excess N fertilizer. Adding more N increases crop yield only up to a point, after which the crop's need for N is saturated and further fertilization has no effect on production (NRC 2000). Land grant universities in the United States advise farmers on appropriate rates of fertilizer application for optimum economic return from crop production under local conditions. In practice, the average farmer in the upper mid-western "breadbasket" area of the United States applies significantly more synthetic N fertilizer than recommended, and 20–30% more than is required to support present crop yields. (See Figure 9.5.) The reasons include underestimation of N available from other sources, such as residues from previous crops, overly optimistic yield expectations, the relatively low cost of N fertilizer, and a tendency to apply extra fertilizer as "insurance" to guarantee maximal yield (Boesch et al. Submitted; Howarth et al. 2002a; Howarth, in press). Note that in some regions with insufficient regulation of N use, farmers overfertilize simply as a way to dispose of animal wastes (discussed below).

Nitrogen not taken up by the crop is available for leaching to surface water and groundwater, and this increases rapidly if N inputs increase beyond the point of crop N saturation. Thus reducing fertilizer use by 20–30% would, in all likelihood, reduce the downstream N pollution by considerably more than 20–30% (Boesch et al., submitted; Howarth et al. 2002a; McIsaac et al. 2001). Such a reduction would also save farmers money, as they would get essentially the same yield but pay less for fertilizer. One promising approach to achieve this goal is the use of voluntary crop production insurance (Howarth, in press). In a trial plan run by the American Farmland Trust, farmers pay into a not-for-profit insurance fund and agree to use less N fertilizer on most of their cropland. Their payments into the fund are less than the savings from purchasing less fertilizer, so the farmers have an economic incentive to participate. Small patches of the fields are heavily fertilized, and the average yield for the entire field planted is compared with the yield in the heavily fertilized plots. If the average yield is below that of the test plots, the farmer is compensated for this lower yield.

Another promising approach for reducing N losses from agriculture fields is the use of precision agriculture, where the timing and amount of fertilization are closely matched to crop needs at relatively small spatial scales (NRC 1993b). Also, genetic engineering may hold promise for increasing the nutrient use efficiency of crops. However, as noted above, significant reduction in nutrient leakage from agroecosystems can be made with existing crops and techniques through such techniques as changing cropping systems, reducing fertilizer use, and employing cover crops.

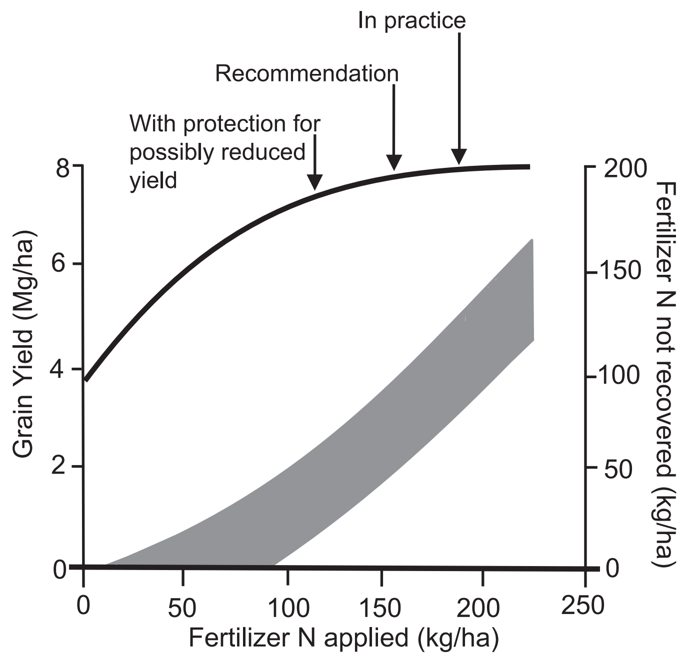

Figure 9.5. Crop Production and Leaching of Nitrogen to Surface and Ground Waters as a Function of Inputs to Agricultural Fields. "Recommendation" indicates application recommended by Land Grant universities based on optimum economic return to farmers. "In practice" indicates the actual average application by farmers. Less fertilizer than the recommended rate for maximizing economic return could be used, with great gains in reducing nitrogen leakage and only small decreases in crop yield. This figure is based on a compendium of real data for farm recommendations and crop production for the upper midwestern United States, where corn and soybean grown in rotation dominate as the cropping system, as of the mid-1990s. (Boesch et al. submitted)

Some agricultural lands in many regions, including the Netherlands and portions of the "breadbasket" midwest in the United States, are artificially drained with tile drains. This is necessary for the growth of most row crops, but it increases leakage of N from the fields. It may be possible to lessen this N leakage by raising the level of drains in fields, while still providing adequate drainage for crop production (Boesch et al., submitted). It is also possible to build artificial wetlands that intercept the tile drainage, provided this does not cause the tiled fields to become flooded. Such wetlands can substantially reduce the flux of nitrate to surface waters (Mitsch et al. 2001). Buffer strips, while effective for trapping P (which is largely particle bound), are not good at trapping N from drainage systems unless the drained water is fully intercepted by the buffer. The subject of artificial or reconstructed wetlands for reducing the flux of nutrients from the landscape is discussed below.

9.4.2 Animal Production and Concentrated Animal Feeding Operations

Animal wastes are a major source of nutrients in many regions, including coastal North Carolina in the United States and many areas of Western Europe, including Brittany, the Netherlands, and

Denmark. In the United States, where per capita meat consumption is among the highest in the world, over half the country's crop production is fed to animals, mostly in feedlots. Most of these crops are transported over long distances before being fed to the animals, making it expensive for the farmers to return the animal wastes to the site of the original crop production (NRC 2000; Howarth et al. 2002a). Instead it is far cheaper for farmers to purchase synthetic fertilizers to use on their fields. The production of animal protein in the United States continues to increase, in part driven by a steady increase in the per capita meat consumption of Americans (Howarth et al. 2002a). The trend for production to concentrate in fewer but larger facilities also continues. During the 1990s, production of hogs, dairy cows, poultry, and beef cattle all rose while the number of operations in each of these segments declined (NRC 2000).

Wastes from concentrated animal feeding operations are normally either spread on agricultural fields, or just held in lagoons. Some operations are beginning to compost animal wastes (NRC 2000). Animal manure can, of course, be used as fertilizer, and recycling it back to agricultural fields is desirable. In practice, however, it is difficult to apply manure at the time and rate needed by the crop, due to the uncertainty about the time of nutrient release and the difficulty of spreading it uniformly (NRC 2000). Most manure is transported only very short distances due to the expense of transporting the heavy waste and the availability of relatively inexpensive synthetic and inorganic nutrient fertilizers, making the use of manure unnecessary for crop production. This results in overfertilization of fields near animal feeding operations, and pollution of ground water and downstream aquatic ecosystems (NRC 2000).

Lagoons are a problematic approach for handling waste, due to loss during flood events, significant leakage of N to ground water, and much volatilization of N as ammonia to the atmosphere. The ammonia contributes to acid rain, the production of fine particles in the atmosphere, loss of biotic diversity in forests and grasslands (Vitousek et al. 1997), and, eventually, the flux of N to coastal waters (NRC 2000). An estimated 40% of all the N in animal wastes in the United States—whether spread on fields as manure or held in lagoons—is volatilized to the atmosphere (Howarth et al. 2002a). In Europe, animal manure is the largest source of atmospheric ammonia emissions, followed by fertilizer use. Estimates of atmospheric ammonia emissions in Europe have considerable uncertainties, but indicate a decrease by about 14% between 1990 and 1998, largely due to decreased agricultural activities in Eastern Europe after 1989. No further decrease is expected by 2010 (Erisman et al. 2003).

Animal wastes can be composted to make them easier for use as effective fertilizers. However, much ammonia is volatilized to the atmosphere during the composting, which lowers the value of the compost as fertilizer and contributes to pollution by atmospheric deposition (NRC 2000). More effective and less polluting methods for treating animal wastes are an urgent need (NRC 2000).

Some progress is being made in developing more environmentally benign approaches for animal wastes, as indicated by the proceedings of a recent symposium sponsored by the International Water Association in Seoul, Korea, on approaches for dealing with nitrogen-rich wastes, including animal wastes. The 135 papers show a wide range of creative and potentially useful approaches, including more effective agricultural re-use and production of biogas for fuel (Choi and Yun 2003). In the European Union, some dairy operations now make more money from biogas production from cow wastes than from selling milk, when the biogas subsidies are considered (Holm Tiessen, personal communication).

Van Asseldonk (1994) showed that environmental efficiency assessments of manure surpluses can be useful to: (1) demonstrate the financial benefits of improved environmental efficiency; (2) influence farmers' attitudes toward the environment; and (3) get farmers more involved in thinking about environmental targets. Leneman et al. (1993) distinguished three main categories, using the handling of pig waste as an example: the first one aims to reduce N and P excretion by changes in feeding and composition regime of the pigs; the second aims to reduce N volatilization and can be carried out in pig houses and on manure storage outside the pig house; finally, the third category aims to reduce N leaching and P leaching in the soil. Combinations of these measures can dramatically reduce N and P emissions (for example, P emissions were reduced as much as 97% in furrowing operations and 95% in finishing operations, and N leaching was reduced by 67% in furrowing operations and by 73% in finishing operations).

The redistribution of manure from areas of high livestock density to arable farming areas did take off to some extent in the Netherlands. Manure processing has turned out to be far more expensive than had been initially thought because of the high cost of developing new technology. Research on manure processing continues, however, particularly in the private sector, because, since 1998, high charges have been imposed on N and P surpluses at the farm level.

9.4.3 Fossil Fuel Sources

As Figure 9.1 shows, the emission of oxidized forms of N (NO_x) to the atmosphere from fossil-fuel combustion contributes approximately 22 teragrams of N to the global environment every year, or roughly 20% of the rate of synthetic N fertilizer use. In China, NO_x emissions, as late as the 1990s, equaled 20% of the use of synthetic N fertilizer, but the increase in NO_x emissions is even greater than that for fertilizer use (shown in Figure 9.4). In the United States, which has the higher per capita emissions of NO_x than any other nation on Earth, the emission rate is 7 teragrams N per year, or 60% of the rate of N fertilizer use in that country (Howarth et al. 2002a). Most NO_x emissions are deposited onto the landscape as rain and as dry deposition, and are the major contributors of acid rain, as well as being significant contributors of the nutrient pollution of coastal waters. A significant percentage of this deposition is exported from forests and other terrestrial systems to rivers and downstream coastal marine ecosystems (NRC 2000; Howarth et al. 2002b), particularly when total N deposition exceeds 8 to 10 kilograms N per hectare per year (Emmet et al. 1998; Aber et al. 2003). Although in most regions of the world N fluxes to the coast are dominated by agricultural sources and human wastewater, in some regions (notably the northeastern United States) atmospheric deposition is the single largest source. Atmospheric deposition of N from fossil-fuel combustion exported via watersheds is a major input to almost all coastal rivers and bays along the eastern seaboard of the United States (NRC 2000; Howarth et al. 2002b; Boyer et al. 2002).

NO_x is the only major pollutant among those in the United States that are regulated under the Clean Air Act that has not declined significantly since the Act was passed in 1970, although regulation may have stabilized the emissions (NRC 2000). Emissions rose exponentially through the 1960s and 1970s, but have been relatively constant since 1980 (EPA 2000), with about half coming from mobile sources, including automobiles, buses, trucks, and off-road vehicles and 42 per cent from electric power generation (EPA 2000). Major sources of NO_x emissions in Europe are transport, industry, and energy production. Work under the Convention on Long-range Transboundary Air Pollution resulted in a reduction of emissions in Europe (Russia excluded) by 21% between 1990 and 1998, mostly due to decreased industrial activity in Eastern Europe. Under the Convention's new Gothenburg Protocol and new EC Directives, reduction is predicted to reach 50% of 1990 emissions by 2010 (Erisman et al. 2003).

Technical controls of NO_x emissions have been much studied because they are central to the formation of ground-level ozone and also contribute to acid rain (NRC 2000; Mosier et al. 2001). The basic approaches are either to burn less fossil- fuel, through greater energy efficiency and reduced driving and/or to remove NO_x from the exhaust as with catalytic converters. NO_x emissions from fossil-fuel combustion in the United States can be almost eliminated with currently available technology (Moomaw 2002). Taking old, ''grandfathered'' power plants off line is a significant and inexpensive step in this direction. Stricter emission standards for sport utility vehicles, trucks, and off-road vehicles are other significant steps. Electric power generation by fuel cells rather than traditional combustion could completely eliminate NO_x emissions from that source (Moomaw 2002).

Efforts to regulate NO_x emissions have been driven largely by the ozone and air quality problems in Europe and North America. The contribution of NO_x to coastal N pollution is a reason for greater efforts, and underlines the need for year-round reductions in emissions; since ozone is a problem mostly in warm weather, current regulatory requirements often focus only on reducing emissions in the summer.

9.4.4 Urban and Suburban Sources

While non-point sources dominate inputs of N to most regions of the world, human wastes are the major source in some regions (such as the Republic of Korea) and are often the largest source to urban estuaries. Human wastes are the primary urban source of N, but atmospheric deposition of N from fossil-fuel combustion can also be substantial, since most NO_x emissions are deposited near emission sources, which are huge in urban areas (Holland et al. 1999; NRC 2000; Howarth et al. 2002b, 2004). In the older cities of both Europe and North America, sanitary wastes and storm waters are mostly combined in the same sewer system, and some of the N entering the sewage treatment plants is, therefore, derived from atmospheric deposition washed off the streets during rainstorms (NRC 2000). Currently, about half of the population of the world lives in urban areas, and the global urban population is expected to continue to grow over this century by 2% per year (UNEP 2002; Austin et al. 2003).

Most human sewage in the world enters surface water with no treatment. Thus while in North America 90% of urban wastewater is treated and in Europe 66%, in Asia only 35% of it is treated, in Latin America and the Caribbean only 14%, and in Africa it is not treated at all (Martinelli 2003). Even in North America, most sewage treatment is not aimed at nutrient reductions, but rather at the reduction of the labile organic matter that contributes to ''biological oxygen demand'' (NRC 2000). This ''secondary treatment'' is not very effective at removing N and P, and the nutrient content of the effluent from an average secondary sewage plant is substantial (NRC 1993a).

Advanced tertiary treatment of sewage for removal of nutrients will, on average, remove up to 90–95% of the P in untreated sewage. Not all tertiary treatment is effective at reducing N, but some technologies can remove up to 90% of the N in sewage. The cost for such tertiary treatment is approximately 25% above that for secondary treatment, including both capital and operating costs (NRC 1993a). For a large urban city such as New York, the additional cost is $30–60 per person per year (Howarth et al. 2004). Substantial savings could be achieved if wastewater treatment plants were designed from the start with the goal of nutrient reduction (NRC 1993a).

9.4.5 Wetlands as Nutrient Interceptors: Enhancing the Sinks

The sections above discussed approaches for reducing anthropogenic nutrient inputs to the environment. Another, complementary strategy is to enhance sinks for nutrients in the landscape. Wetlands, ponds, and riparian zones are particularly effective nutrient traps, serving both to sediment out particulate forms of N and P and to convert biologically available combined N into N_2 and N_2O gases through the process of denitrification (NRC 2000; Howarth et al. 1996; Mitsch et al. 2001). N_2 is a ubiquitous and harmless gas but, unfortunately, N_2O is a very long-lived gas in the atmosphere (about 120 years) that contributes both to climate change and to the creation of ozone holes in the stratosphere (Vitousek et al. 1997; Howarth 2002). Mitsch et al. (2001) estimate that the N load from the heavily polluted Illinois River basin to the Mississippi River could be cut in half by converting 7% of the basin back to wetlands. Restoration of water flows through the wetlands of the Mississippi River delta could also significantly reduce N fluxes onto the continental shelf, where they contribute to the Gulf's hypoxic zone (Mitsch et al. 2001; Boesch et al., submitted). Small natural streams can also be extremely effective sinks (Peterson et al. 2001).

A major unknown is whether the amount of N_2O that is produced during denitrification in wetlands and small streams is greater than or less than the amount that would be produced during denitrification in downstream coastal marine ecosystems (including on the continental shelf), which would otherwise be the fate of much of this excess N (Nixon et al. 1996). A major research goal should be to evaluate this trade-off, and to work towards designing wetlands as treatment systems that minimize the production of N_2O (Howarth et al. 2003).

9.5 Analysis and Assessment of Selected Responses

Most nations of the developed world have responded to the problem of excesses of P and N in surface waters and in the air with legislative and regulatory responses. However, these often have not proven adequate, particularly for N pollution. In this section, the various types of responses that have been proposed or tried are reviewed, and their potential usefulness is assessed. For the most part, developing-country governments have not yet managed nutrient pollution. Thus the focus of the discussion here is of necessity on the industrial world, particularly the United States but also some countries in Western Europe.

9.5.1 Watershed-based versus Nationally Uniform Approaches

The United States needs a national strategy to reduce N pollution in coastal waters (NRC 2000). Federal involvement is appropriate, because the nutrients polluting many coastal ecosystems come from large river systems that flow through many states and, for N, from large multistate air sheds. Notable examples include Long Island Sound, Chesapeake Bay, and the Mississippi River plume. Further, national agricultural policies significantly affect N pollution. However, the problem of N pollution manifests itself at the local to regional scale, so local and state governments also clearly have a role. Since coastal ecosystems vary both in sensitivity to N pollution and in their sources of N, a national goal of protecting ecosystems not yet damaged and of restoring those that have been damaged seems preferable to a goal of N reduction per se (NRC 2000; Howarth, in press). This would require a partnership of federal, state, and local authorities, cooperating with academia and industry.

This approach posits that the coastal ecosystems that should be restored first are those that are most sensitive to nutrient pollution, that have nutrient sources that can most effectively and economically be reduced, or that have the greatest ecological or societal value. The watershed (and associated air shed) is viewed as the appropriate scale for management. A benefit of this approach is that by targeting locations, limited technical and financial resources are most effectively used.

A watershed-specific approach requires estimates of maximum allowable nutrient loads to coastal rivers and bays. As a start, this could be based on the average responses of these ecosystems to increased inputs of N, which is most often the limiting nutrient (NRC 1993a, 2000; Nixon 1995). Since coastal marine ecosystems vary in their sensitivity to nutrient inputs, however, the most cost-effective protection would be obtained by setting higher allowable N load limits for ecosystems that are insensitive to N inputs and lower limits for systems that are highly sensitive (NRC 2000). This would require that coastal rivers and bays are classified according to their sensitivity to nutrient pollution and that loading limits be established by this classification (NRC 2000). An alternative but more time-consuming option is to construct site-specific models for each coastal river and bay (NRC 2000). Note that unlike many pollutants, concentrations of N are a poor predictor of the effects of N pollution in the coastal zone. A strong scientific consensus exists that N pollution should be managed on the basis of N input rates (loads) (NRC 2000). The "total maximum daily load" provision of the Clean Water Act of the United States (discussed further below) is a regulatory mechanism that uses allowable loads for reducing nutrient pollution.

A watershed-specific approach is technically challenging, since the ability to classify coastal ecosystems as to their sensitivity to nutrient pollution is lacking except in broad outline (NRC 2000). Also lacking is detailed, reliable knowledge on the sources of N to most individual coastal ecosystems, although the general patterns at the regional or national scale are clear (NRC 2000; Howarth et al. 2002b). A major strategy of the Clean Water Action Plan put forward in 1998 was to develop nutrient criteria that could aid watershed management, but nutrient criteria for coastal waters are still many years away, as discussed below. While current knowledge is sufficient for starting to restore individual coastal rivers and bays, more research on the sensitivity of ecosystems to nutrient pollution and on sources of nutrients to individual ecosystems will be needed to find cost-effective solutions (NRC 2000; Howarth et al. 2003).

The European Union decided in 2001 to adopt a mainly watershed-based approach (the "Water Framework Directive") to water quality management for its groundwater, fresh water, and coastal water (Chave 2001). The approach is a mixed one, with earlier directives, such as the technology-based "Urban Wastewater Treatment Directive" and the "Nitrate Directive" for managing agricultural N pollution, initially remaining in force, but eventually being replaced by the Water Directive. The stated objective is to achieve good, or better, water status for all covered waters by 2015.

An alternative to the watershed-based approach would be a uniform national regulation to reduce, for example, overall N fluxes to the coastal waters of the United States, by 10% by 2010 and by 25% by 2020, without regard to the effects on individual coastal ecosystems. A uniform approach requires less technical expertise and site-specific information than a watershed-specific approach. If the reductions in N flow occur to coastal systems that are insensitive to N pollution, however, they will bring little, if any, environmental benefit. Moreover, increased local loads to some sensitive areas cannot be ruled out. Consequently, this approach is likely to be less cost-effective, and may fail to protect the most sensitive coastal ecosystems (NRC 2000).

9.5.2 Voluntary Policies for Reaching Goals

Both voluntary and mandatory approaches have been used for nutrient management, and both should be considered as part of the national strategies against nutrient pollution, whether these strategies use a watershed-specific or a uniform national approach. Motivations for polluters to join a pollution abatement plan voluntarily include a commitment to environmental stewardship, a perceived payoff in the marketplace (selling a "green" product), a financial incentive or subsidy, and a fear that failure to participate will lead to stricter regulatory control (NRC 2000). The regulatory threat as a powerful motivator for voluntary compliance is an argument for hybrid approaches, which combine regulations and voluntary programs (NRC 2000).

Voluntary approaches have been used successfully to reduce N pollution in Tampa Bay. On the other hand, after trying voluntary nutrient management on farms, the State of Maryland, in 1998, moved to mandatory control (NRC 2000). The Integrated Assessment on Hypoxia in the Gulf of Mexico brought together a diverse group of scientists from governments and academia to produce consensus reports on the problem of the hypoxic zone, and on approaches for solving the problem. These reports formed the basis for negotiations between the federal government and the state governments regarding the Mississippi River basin. These resulted, as a first step, in a voluntary agreement in 2001 to reduce the size of the hypoxic zone by reducing N loading down the Mississippi River over the next few decades. Whether voluntary cooperation will prove sufficient to reach this goal remains to be seen, but the Integrated Assessment has proven that voluntary steps can be taken even at large, multistate scales.

Robinson and Napier (2002) studied the adoption of nutrient management techniques to reduce hypoxia in the Gulf of Mexico. They collected data from 1,011 landowner-operators within the three watersheds located in the north-central region of the United States to examine use of selected water protection practices; their research findings suggest that existing conservation programs are no longer useful policy instruments for motivating landowner-operators to adopt and use production systems designed to reduce agricultural pollution of waterways. They strongly suggest that policy-makers should reconsider allocation of limited funding for conservation-education and for efforts designed to increase access of farmers to training.

Financial incentives and subsidies can facilitate voluntary solutions. Incentive programs, however, risk distorting the economy in counter-productive ways. If a firm is subsidized to reduce the discharge of a pollutant, its costs and the price of its products can be kept artificially low. This can increase demand for the product, and encourage other firms to enter the industry, so that pollution actually increases (NRC 2000).

Economic incentives are a long-established part of farm policy both in the United States and the European Union, and are based on providing technical assistance and subsidies, including policies for reducing pollution from farms (NRC 2000). The Conservation Reserve Program of the United States has successfully reduced erosion and increased habitat for wildlife in the agricultural landscape through financial payments to farmers who take land out of agricultural production and create buffer strips around streams. This program is not designed to reduce N pollution, but financial incentives could clearly also be used to encourage farmers to undertake best-management practices for N reduction (NRC 2000).

A look at the use of fertilizer subsidies and their removal throughout sub-Saharan Africa in the 1990s as part of structural adjustment programs presents some useful lessons. Removal of subsidies in many cases has led to the reduction in fertilizer use but this has not been so where prices for crops increased more than that of inputs (Scoones and Toulmin 1999). Other cases, however, show that even with large subsidies for fertilizers, there was still little use; and use may be more related to availability than price (Manyong et al. 2002). In any event, throughout much of sub-Saharan Africa, current fertilizer prices are two to six times those in most other places (Sanchez 2002). Even if the prices had been at par the majority of farmers would not have been able to afford the amounts needed to raise yields sufficiently nor would it have been cost-effective even if yields had been raised substantially given relative crop prices. Although a thorough analysis has not been done, transportation and boundary-crossing costs are, most likely, two of the main reasons for these excessive costs. Efforts should be made to address these issues. Improved road infrastructure will also increase access to markets and shift relative prices and, perhaps, provide more incentive for soil fertility replenishment (Scoones and Toulmin 1999).

There has been considerable debate regarding "subsidies" (or public intervention) for soil fertility replenishment (Sanchez et al. 1997; Scoones and Toulmin 1999). Where policy distortions affect input and/or output markets and where soil nutrient depletion affects livelihoods, then public interventions may be appropriate (Scoones and Toulmin 1999). Indeed, farmers will only invest in soil fertility management if there is a perceived benefit. Even then they may not have the labor and capital to do so, and there are competing demands (education and health) for the scarce capital they do have. Scoones and Toulmin (1999) conclude that there are grounds for public intervention to intensify sustainable agriculture in sub-Saharan Africa given the importance of agriculture for providing food, incomes, and employment.

9.5.3 Mandatory Policies for Reaching Goals: Regulations

9.5.3.1 Technology-based Standards

Mandatory policies, including regulatory control and tax or fee systems, place the costs and burden of pollution control on those who generate the pollution (NRC 2000). Technology-based standards are easy to implement but tend to discourage innovation and are generally not seen as cost-effective (NRC 1993a, 2000). In the United States, regulation under the Clean Water Act has been largely technology-based since its inception (Powell 2001). Since 1977, the Clean Water Act requires point sources of pollution to meet technology-based standards, administered by the Environmental Protection Agency (Powell 2001). For publicly owned sewage treatment plants, the standard remains secondary treatment (NRC 1993a; Powell 2001), designed merely to reduce the discharge of pathogens and labile organic matter, generally referred to in terms of the biological oxygen demands (BOD) created from this organic matter, and is rather ineffective at removing nutrients (NRC 1993a). Tightening the technology standard to include effective nutrient removal could substantially reduce N pollution at modest cost (NRC 1993a, 2000). Communities that can demonstrate that a lower level of treatment results in no significant environmental deterioration could obtain waivers from the standard from EPA (NRC 1993a).

Animal feeding operations are subject to the requirements of the Clean Water Act, but compliance has been poor, with only 20% of CAFOs having the necessary permits (Powell 2001). Currently, operations with more than 1,000 "animal units" are prohibited from discharging to surface waters, except during overflow conditions expected during a 25-year storm (Powell 2001). New EPA proposed new regulations for effluents were signed in December 2002 and published in the *Federal Register* in February 2003. While the intent was to regulate land application of manure as well as lagoon systems, the new regulations contain some provisions for alternative technologies, including an option to develop an alternative technology that considers pollution losses by all media (air, surface water, ground water) and results in a net reduction (EPA 2003). Volatilization of ammonia to the atmosphere would, however, remain unregulated (Powell 2001).

9.5.3.2 Total Maximum Daily Loads

The Clean Water Act requires states to monitor for violations of ambient water quality standards and, when a standard is violated, to determine the "total maximum daily load" that could enter the water body without causing impairment. If the TMDL is exceeded, the discharges allowed from point sources are lowered in steps (Powell 2001). However, non-point sources dominate N input to most coastal waters (NRC 2000), and the Clean Water Act provides no authority for regulating non-point sources in the TMDL context (Powell 2001). New statutory authority will be required for non-point sources, if N pollution is to be reduced through regulations.

Since 1990, states participating in the Coastal Zone Management Program of the United States are required to have enforceable mechanisms for controlling non-point source pollution. In many states the coastal zone is too narrowly defined, however, for effective nutrient management. Further, many states fail to comply with this requirement, and federal agencies have no authority to force compliance (Powell 2001). Funding for the Coastal Zone Management Program is small, giving federal agencies little leverage over state actions.

Under current U.S. law, TMDLs are applied by a state based on compliance with water quality standards within that state. This is problematic when pollution from one state contributes to impairment of a water body in another state. For example, N coming down the Susquehanna River from Pennsylvania is a major source of pollution to Chesapeake Bay (Virginia and Maryland), but this has not influenced TMDLs within Pennsylvania. If the TMDL regulatory approach is to be successful in reducing coastal N pollution, not only is new authority required for non-point source pollution, but also multiple state sources must be included. Providing enforcement authority to river basin commissions or other similar watershed-based entities may help in achieving this goal.

Although long mandated by the Clean Water Act, the TMDL approach has only been applied recently, after litigation led federal courts to direct the EPA to develop TMDLs. Political opposition remains vocal, which led Congress to ask the National Research Council to assess the scientific basis for TMDLs. The appointed committee endorsed the basic usefulness of TMDLs and suggested ways in which their usefulness could be improved, such as explicitly recognizing uncertainty and relying more on biological endpoints for standards (NRC 2001).

The TMDL approach is based on water quality standards. Currently states either lack nutrient standards for coastal waters or have only loose narrative standards (NRC 2000). EPA is working to develop procedures that could be used by the states to set nutrient standards, and has published the *Nutrient Criteria Technical Guidance Manual for Estuarine and Coastal Marine Waters* (EPA 2001a). States are expected to develop nutrient standards for fresh waters by 2004 (Powell 2001), but no deadline has been set for coastal marine ecosystems. Meanwhile, TMDLs for N control are sometimes driven by other standards, for example, in Long Island Sound the current plan for reducing N pollution is designed to comply with the local dissolved oxygen standard (NRC 2000).

9.5.4 Mandatory Policies for Reaching Goals: Taxes, Fees, and Marketable Permits

Mandatory taxes and fees that could be used as regulatory instruments for inducing change include effluent charges, user or product charges, non-compliance fees, performance bonds, and legal liability for environmental damage (NRC 2000). These approaches are widely believed to be more cost-effective than command-and-control regulations and to be more likely to spur innovation. For example, gasoline taxes could be increased to reduce fuel use and, hence, NO_x emissions, and N fertilizer could be taxed to reduce its use and to encourage appropriate use of manure. It is, however, difficult to reach specific targets in pollution reduction using the tax/fee approach (NRC 2000), since regulators have difficulty predicting how polluters will react. Adjusting fees and taxes over time to achieve the desired result may create political resistance, and the rates of taxation necessary to bring about meaningful changes in behavior are likely to be seen as excessive by large segments of the public (NRC 2000).

Marketable permits for pollution avoid some of the problems of both regulation and tax/fee systems and have been used to reduce sulfur dioxide pollution from electric power plants (NRC 2000). As with the regulatory approach, marketable permits start by deciding on an allowable level of pollution, so there is an assurance that the set target will be met. By allowing trading among permit holders, however, innovation is encouraged and the most economic abatement is achieved, given a sufficient number of buyers and sellers (NRC 2000). Marketable permits are now being tried for N control to coastal waters in several locations, including Pamlico Sound (North Carolina) and Long Island Sound. To date no trading has actually occurred. A major obstacle is that the need to establish a basis for trading between point and non-point sources of pollution requires precise knowledge on the sources and extent of non-point pollution (Malik et al. 1993; NRC 2000). This is seldom available for watersheds, although the ability of models to assess sources is improving rapidly (NRC 2000).

The EPA plans to use market-based trading programs for future reduction of point and non-point source nutrient pollution in U.S. waters, with trading occurring within watersheds (EPA 2001b). The program would have caps for total nutrient pollution based on water quality standards and for impaired waters on TMDLs. The proposed trading program is thought to provide economic incentives for voluntary reductions from non-point pollution sources, and would retain permit requirements for point source pollution (EPA 2001b). The lack of statutory authority for regulating non-point sources of nutrients under the Clean Water Act (Powell 2001) will be a major problem for the plan in the majority of watersheds, where non-point sources dominate N input. The lack of authority over inter-state pollution is another challenge.

9.5.5 Hybrid Approaches for Reducing Coastal Nitrogen Pollution

Nitrogen pollution has multiple sources, and an effective national strategy may require a combination of national regulation of some sources and watershed-based management of others. Combinations of regulatory, incentive, and market-based mechanisms are possible for both national and watershed-based approaches and may be the most cost-effective and politically acceptable (NRC 2000; Howarth, in press). A brief discussion of hybrid policy options for N pollution follows.

9.5.5.1 Runoff and Leaching from Agricultural Fields

Leakage of N from farming could be substantially reduced if farmers fertilized at, or even somewhat below, the rate recommended for maximizing profit. To achieve this, national farm policy should improve the economic return of those farmers who appropriately reduce fertilizer use and reduce economic subsidies to those farmers who exceed recommended fertilization levels. The crop-insurance program of the NGO American Farmland Trust demonstrates an approach that may work when governmental leadership is lacking (Howarth, in press).

Other means of reducing N loss include winter cover crops, planting perennial rather than annual crops, discouraging fall application of fertilizer, and using wetlands to intercept farmland drainage. Any of these could be encouraged, nationally or in specific watersheds, through incentive payments. Regulations or marketable permits that charge farmers for N runoff above a set limit could also be applied. Within watersheds, either incentive systems or regulatory and market-based approaches could be tailored to the largest problems, such as farmers who grow annual row crops such as corn and soybeans on sandy soils in a wet climate.

9.5.5.2 Concentrated Animal Feeding Operations

The scale of pollution from CAFOs is nearly equal to that from municipal sewage treatment plants, and it might be sensible to apply similar national technology-based standards to CAFOs as have been applied to point source pollution under the U.S. Clean Water Act. On the other hand, a performance-based standard may encourage more innovation in treatment technologies and therefore be less expensive to the industry in the long run (NRC 1993a, 2000). Marketable permits could also be used to control pollution from CAFOs. In any case, release of nutrients to surface and ground waters and to the atmosphere should all be controlled. The fate of manure from CAFOs must also be considered. In the Netherlands and a few other countries of Europe, manure application to fields is now regulated as part of the total farm nutrient-balance programs (NRC 2000), as has also been recently mandated in Nebraska and Maryland (NRC 2000; Mosier et al. 2001). The goal is to prevent overfertilization as a disposal mechanism for the manure.

Several European countries with high concentrations of livestock have their own manure regulations. In all countries, the quality of drinking water and surface water is an important policy issue, particularly with respect to N levels. Command-and-control measures are the most common policy tools for manure regulation in the countries of the European Union. In the Netherlands, for example, economic incentives have been used since 1998 in the form of an economically significant tax on N and P surpluses at farm level. See Breembroek et al. (1996) for details on the information system to assess these surpluses.

9.5.5.3 NO_x from Fossil Fuel Combustion

Atmospheric NO_x pollution comes from multiple sources, including on- and off-road vehicles and stationary sources such as electric power plants, and can be transported over long distances, so that the airshed for pollution sources usually overlaps several countries and watersheds (NRC 2000). This suggests that NO_x emissions are, perhaps, best regulated at the national scale. Hybrid regulatory approaches are possible at the national level, for instance, relying on emission standards for vehicles and marketable permits for electric power plants. Coastal N pollution may require year-round standards rather than the summer-time standards commonly used for control of ozone and smog pollution (NRC 2000).

9.5.5.4 Urban and Suburban Sources

Municipal sewage treatment plants are currently regulated by a technology-based standard (secondary treatment) and are not related to the degree of nutrient pollution. Upgrading the national technology standard from secondary treatment to nutrient removal may be sensible, and the cost is moderate (see discussion above). Where upgrading would serve no purpose, municipalities could apply for waivers under the Clean Water Act, as they currently can for the secondary sewage treatment standard (NRC 1993a, 2000).

Another approach would be to require nutrient removal only in plants located in watersheds that contribute to known N pollution in downstream coastal rivers or bays. Currently, N-reduction technology is required on a case-by-case basis and that too only when sewage plants discharge directly into coastal ecosystems. Nitrogen discharges elsewhere in river basins have largely escaped regulation.

9.5.5.5 Wetland Creation and Preservation

Wetlands and ponds can be significant sinks of both N and P. Wetlands and ponds that directly intercept groundwater flows or tile drainage, or have effective water exchange with streams and rivers, are the greatest sinks of N, while those that intercept eroding sediments are most effective for P (Howarth, in press). Incentive programs could target the creation and preservation of wetlands designed to be good N sinks, in watersheds with N pollution problems. Federal water management programs could significantly influence the ability of wetlands and floodplains to serve as N sinks, and some authors have strongly urged that increasing N retention should be made a central concern in the planning of such projects (Mitsch et al. 1999, 2001). However, wetlands can release substantial quantities of greenhouse gases (methane, in addition to N_2O), and this should be carefully balanced against the increased nutrient sinks when planning any large-scale strategy for increased wetlands.

9.6 Lessons Learned and Synthesis

Excess nutrients cause grave pollution problems in many rivers, lakes, waterways and coastal rivers and bays in the developed countries and now, increasingly, also in the developing ones. In the United States, two thirds of all coastal systems are moderately to severely degraded by nutrient pollution.

Nutrient inputs to inland and coastal waters are increasing globally. In some developed countries, discharges of P have been effectively curtailed, but N pollution has been reduced only in some local areas. If current trends continue, N loading to the coastal waters of the United States in 2030 is projected to be 30% above what it is today and more than twice that what it was in 1960, and increases are likely to be even larger in developing countries with rapid population growth.

The strategies for replenishing the immediate N supply in the soils for crop uptake are not necessarily compatible with replenishing the longer-term storage of N in the soil organic matter (Palm et al. 2001). Fairly small amounts of high-quality organic inputs can satisfy the immediate N demand of crops but does little to build the soil organic N (Giller et al. 1997; Merckx 2002). Increasing soil organic N would be best accomplished with systems that combine large amounts of organic inputs, such as legume fallows, and minimum tillage, and even then continued applications of N to balance crop demands would be necessary (Giller et al. 1997). These systems should be designed to mitigate the potential extra losses from pooled N.

The nutrients come from a variety of sources, including agricultural fields, concentrated animal feeding operations, sewage and septic wastes, and, for N, atmospheric deposition of NO_x from fossil-fuel combustion. The relative importance of these varies among receiving water ecosystems.

Technical tools and management practices exist for reduction of nutrient pollution at reasonable cost. That many of these have not yet been implemented on a significant scale suggests that new policy approaches are needed.

The best management solution may often be a combination of some of the many voluntary and mandatory approaches available, perhaps applying different approaches to different sources of nutrient pollution.

Non-point sources dominate N inputs to most coastal waters, and are important also for P. Current regulatory authority for non-point source pollution is often nonexistent or very limited. Hence, increased authority to regulate such sources may be necessary to reverse pollution of surface waters by nutrients.

Nutrient pollution can be addressed through a uniform national approach, or a watershed-based approach, or through a combination of these (for example, by applying a uniform national approach to NO_x emissions, while also setting watershed-based N loading standards). A watershed-based approach is likely to be the most cost-effective for some sources of nutrients (such as runoff from agricultural fields), while a uniform national approach may be better for others (such as NO_x from fossil-fuel combustion or P from detergents).

While current scientific and technical knowledge is sufficient to begin making real progress toward eliminating coastal nutrient pollution, progress will be quicker and more cost-effective with increased investment in appropriate scientific research.

Since nutrient pollution in many coastal ecosystems has sources in multiple states, state and local governments may not be the most effective regulatory agencies for a watershed-based approach. River basin commissions or similar entities, if given sufficient authority, may be more appropriate. Economic research on environmental policy design has largely been concerned with the merits of emissions-based economic incentives (for example, emission charges, emission reduction subsidies, transferable discharge permits). This literature has limited relevance to nutrients from agriculture and other non-point sources since emissions in such cases are, for all practical purposes, unobservable and typically stochastic. Several features of non-point pollution problems complicate the identification of good solutions. One is the high degree of uncertainty about non-point emissions and their fate. Given current monitoring technology, non-point emissions attributable to particular decision-makers cannot be measured with reasonable accuracy at reasonable cost. The result is substantial uncertainty about the decision-makers who are responsible for non-point pollution and about the degree of each firm's or household's responsibility. This rules out the use of the kinds of emission-based instruments that economists usually advocate for cost-effective pollution control (Griffin and Bromley 1983; Shortle and Dunn 1986).

Another feature of non-point problems is the extreme spatial variation in the feasibility, effectiveness, and cost of technical options for reducing emissions. This greatly limits the applicability of the uniform technology-based regulatory approaches that are often used to control point sources (Malik et al. 1994). A variety of options are available to encourage farmers to make socially desirable choices. The menu consists of choices about who should comply, how their performance (or compliance) will be measured, and the policy tools that will be used to affect their choices. Each of these choices will affect the economic and ecological performance but problem-specific research is essential for resolving this issues.

References

Aber, J.D., C.L. Goodale, S.V. Ollinger, M.L. Smith, A.H. Magill, et al., 2003: Is nitrogen deposition altering the nitrogen status of northeastern forests? *BioScience,* **53(4),** pp. 375–389.

Austin, A., R.W. Howarth, J.S. Baron, F.S. Chapin, T.R. Christensen, et al., 2003: Human disruption of element interactions: Drivers, consequences, and trends for the 21st Century. In: *Interactions of the Major Biogeochemical Cycles: Global Change and Human Impacts, SCOPE #61,* J.M. Melillo, C.B. Field, and B. Moldan (eds.), Island Press, Washington, DC, pp. 15–45.

Bashkin, V.N., S.U. Park, M.S. Choi, and C.B. Lee, 2002: Nitrogen budgets for the Republic of Korea and the Yellow Sea region, *Biogeochemistry,* **57/58,** pp. 387–403.

Bationo, A., S.K. Mughogho, and A.U. Mokwunye, 1986: Agronomic evaluation of phosphate fertilizers in tropical Africa. In: *Management of Nitrogen and P Fertilizers in Sub-Saharan Africa,* A.U. Mokwunye and P.L.G. Vlek (ed.), Martinus Nijhoff, Dordrecht, The Netherlands, pp. 283–318.

Boesch, D.F., 2002: Reversing nutrient over-enrichment of coastal waters: Challenges and opportunities for science, *Estuaries,* **25,** pp. 744–58.

Boesch, D.F., R.H. Burroughs, J.E Baker, R.P. Mason, C.L. Rowe, et al., 2001a: Marine pollution in the United States: Significant accomplishments, future challenges, PEW Oceans Commission, Arlington, VA.

Boesch, D.F., R.B. Brinsfield, and R.E. Magnien, 2001b: Chesapeake Bay eutrophication: Scientific understanding, ecosystem restoration, and challenges for agriculture, *Journal of Environmental Quality,* **30,** pp. 303–20.

Boesch, D.F., R.B. Brinsfield, R.W. Howarth, J.L. Baker, M.B. David, et al., Submitted to *Improving Water Quality While Maintaining Agricultural Production.*

Bohlool, B.B., J.K. Ladha, D.P. Garrity, and T. George, 1992: Biological nitrogen-fixation for sustainable agriculture: A perspective, *Plant and Soil,* **141(1–2),** pp. 1–11.

Boyer, E.W., C.L. Goodale, N.A. Jaworski, and R.W. Howarth, 2002: Anthropogenic nitrogen sources and relationships to riverine nitrogen export in the northeastern USA, *Biogeochemistry.* In press.

Breembroek, J.A., B. Koole, K.J. Poppe, and G.A.A. Wossink, 1996: Environmental farm accounting: The case of the Dutch nutrient bookkeeping system, *Agricultural Systems,* **51,** pp. 29–40.

Bricker, S.B., C.G. Clement, D.E. Pirhalla, S.P. Orland, and D.G.G. Farrow, 1999: *National Estuarine Eutrophication Assessment: A Summary of Conditions, Historical Trends and Future Outlook,* National Ocean Service, National Oceanic and Atmospheric Administration, Silver Springs, MD.

Buresh, R.J., P.C. Smithson, and D.T. Hellums, 1997: Building soil P capital in Africa. In: *Replenishing Soil Fertility in Africa,* R.J. Buresh, P.A. Sanchez, and F. Calhoun, (eds.), SSSA special publication no. 51, Soil Science Society of America, Madison, Wisconsin, pp. 111–49.

Carpenter, S.R., N.F. Caraco, D.L. Correll, R.W. Howarth, A.N. Sharpley, et al., 1998: Non-point pollution of surface waters with P and nitrogen, *Issues in Ecology,* **3,** pp. 1–12.

Chave, P., 2001: *The EU Water Framework Directive: An Introduction,* IWA Publishing, London, UK, 208 pp.

Choi, E., and Z. Yun (eds.), 2003, *Proceedings of the Strong N and Agro 2003,* Two vols., IWA Specialty Symposium on Strong Nitrogenous and Agro-Wastewater, 11–13 June 2003, Seoul, Korea, 1,118 pp.

Cleveland, C.C., A.R. Townsend, D.S. Schimel, H. Fisher, R.W. Howarth, L.O. Hedin, et al., 1999: Global patterns of terrestrial biological nitrogen (N_2) fixation in natural systems, *Global Biogeochemical Cycles,* **13,** pp. 623–45.

Critchfield, R., 1982: Science and the villager: The last sleeper wakes, *Foreign Affairs,* **61(1)** (Fall).

Dobereiner, J., 1997: Biological nitrogen fixation in the tropics: Social and economic contributions, *Soil Biology and Biochemistry,* **29,** pp. 771–74.

Dobereiner, J., S. Urquigua, and R.M. Boddey, 1995: Alternatives for nitrogen nutrition of crops in tropical agriculture, *Fertilizer Research,* **42(1–3),** pp. 339–46.

Dudal, R., 2002: Forty years of soil fertility work in sub-Saharan Africa. In: *Integrated Plant Nutrient Management in Sub-Saharan Africa: From Concept to Practice,* B. Vanlauwe, J. Diels, N. Sanginga, and R. Merckx (eds.), CAB International, Wallingford, UK, pp. 7–21.

Emmett, B.A., D. Boxman, M. Bredemeir, P. Gunderson, O.J. Kjonaas, et al., 1998: Predicting the effects of atmospheric deposition in conifer stands: Evidence from the NITREX ecosystem scale experiments, *Ecosystems,* **1(4),** pp.352–360

Elmgren, R. and U. Larsson, 2001: Nitrogen and the Baltic Sea: Managing nitrogen in relation to P. In: *Optimizing Nitrogen Management in Food and Energy Production and Environmental Protection,* Proceedings of the 2nd International Nitrogen Conference on Science and Policy, The Scientific World 1 (S2), Berkshire, UK, pp. 371–77.

Erisman, J.W., P. Grennfelt, and M. Sutton, 2003: The European perspective on nitrogen emissions and deposition, *Environment International,* **29,** pp. 311–25.

Foy, R.H, S.D. Lennox, and C.E. Gibson, 2003: Changing perspectives on the importance of urban P inputs as the cause of nutrient enrichment in Lough Neagh, *Science of the Total Environment,* **310,** pp. 87–99.

Galloway, J.N., J.D. Aber, J.W. Erisman, S.P. Seitzinger, R.H. Howarth, 2003: The nitrogen cascade, *BioScience,* **53,** pp. 341–56.

Galloway, J.N., F.J. Dentener, D.G. Capone, E.W. Boyer, R.W. Howarth, et al., 2004: Nitrogen cycles: Past, present, and future, *Biogeochemistry.* In press.

Giller, K.E., G. Cacisch, C. Ehaliotis, E. Adams, W. Sakala., et al., 1997: Building soil nitrogen capital in Africa. In: *Replenishing Soil Fertility in Africa,* R.J. Buresh, P.A. Sanchez, and F. Calhoun, (eds.), SSSA special publication no. 51, Soil Science Society of America, Madison, WI, pp. 151–92.

Goolsby, D.A., W.A. Battaglin, G.B. Lawrence, R.S. Artz, B.T. Aulenbach, et al., 1999: *Flux and Sources of Nutrients in the Mississippi-Atchafalaya River Basin, Topic 3,* Report for the Integrated Assessment on Hypoxia in the Gulf of Mexico, NOAA Coastal Ocean Program Decision Analysis Series No. 17, National Oceanic Atmospheric Association, Silver Spring, MD.

Goolsby, D.A. and W.A. Battaglin, 2001: Long-term changes in concentrations and flux of nitrogen in the Mississippi River basin, USA, *Hydrological Processes,* **15,** pp. 1209–26.

Griffin, R.C. and D.W. Bromley, 1983: Agricultural runoff as a non-point externality: A theoretical development, *American Journal of Agricultural Economics,* **70,** pp. 37–49.

Grimwall, A., P. Stålnacke, and A. Tonderski, 2000: Time scales of nutrient losses from land to sea: A European perspective, *Ecological Engineering,* **14,** pp. 363–71.

Hayami, Y. and K. Otsuka, 1994: Beyond the green revolution: Agricultural development strategy into the new century. In: *Agricultural Technology: Policy Issues for the International Community,* J.R. Anderson. (ed.), CAB International, Wallingford, UK, pp. 15–42.

Holland, E.A., F.J. Dentener, B.H. Braswell, and J. M. Sulzman, 1999: Contemporary and pre-industrial global reactive nitrogen budgets, *Biogeochemistry,* **46,** pp. 7–43.

Howarth, R.W.: *The Development of Policy Approaches for Reducing Nitrogen Pollution to Coastal Waters of the USA,* Proceedings of the 3rd International Nitrogen Symposium, Nanjing, China. In press.

Howarth, R.W., 2002: The nitrogen cycle. In: *Encyclopedia of Global Environmental Change, Vol. 2, The Earth System: Biological and Ecological Dimensions of Global Environmental Change,* H.A. Mooney and J.G. Candell (eds.), Wiley, Chichester, UK, pp. 429–35.

Howarth, R.W., 2003. Human acceleration of the nitrogen cycle: Drivers, consequences, and steps towards solutions. In: E. Choi, and Z. Yun (eds.), Proceedings of the Strong N and Agro 2003 IWA Specialty Symposium, Korea University, Seoul, Korea, pp. 3–12.

Howarth, R.W., H. Jensen, R. Marino, and H. Postma, 1995: Transportation and processing of P in near-shore and oceanic waters. In: *P in the Global Environment, SCOPE #54,* H. Tiessen (ed.), Wiley & Sons, Chichester, UK, pp. 323–45.

Howarth, R.W., G. Billen, D. Swaney, A. Townsend, N. Jaworski, et al., 1996: Regional nitrogen budgets and riverine N and P fluxes for the drainages to the North Atlantic Ocean: Natural and human influences, *Biogeochemistry,* **35,** pp. 75–139.

Howarth, R.W., D. Anderson, J. Cloern, C. Elfring, C. Hopkinson, et al., 2000: Nutrient pollution of coastal rivers, bays, and seas, *Issues in Ecology,* **7,** 1–15.

Howarth, R.W., E.W. Boyer, W.J. Pabich, and J.N. Galloway, 2002a: Nitrogen use in the United States from 1961–2000 and potential future trends, *Ambio,* **31(2),** 88–96.

Howarth, R.W., D. Walker, and A. Sharpley, 2002b: Sources of nitrogen pollution to coastal waters of the United States, *Estuaries,* **25,** pp. 656–76.

Howarth, R.W., R. Marino, and D. Scavia, 2003: Priority topics for nutrient pollution in coastal waters: An integrated national research program for the United States, National Ocean Service, National Oceanic and Atmospheric Administration, Silver Spring, MD.

Howarth, R.W., R. Marino, D.P. Swaney, and E.W. Boyer, 2004: Wastewater and watershed influences on primary productivity and oxygen dynamics in the lower Hudson River estuary. In: *The Hudson River,* J. Levinton (ed.), Academic, New York, NY.

Howarth, R.W. and R. Marino, Nitrogen as the limiting nutrient for eutrophication in coastal marine ecosystems: Evolving views over 3 decades, *Limnology and Oceanography.* In press.

IFA (International Fertilizer Industry), 2002: Fertilizer Indicators, IFA Web page. Available at http://www.fertilizer.org/ifa/statistics.asp.

Insel, Barbara, 1985: A world awash in grain, *Foreign Affairs,* **63(4)** (Spring).

Jaworksi, N.A., R.W. Howarth, and L.J. Hetling, 1997: Atmospheric deposition of nitrogen oxides onto the landscape contributes to coastal eutrophication in the north-east US, *Environmental Science & Technology,* **31,** pp. 1995–2004.

Karl, D., A. Michaels, B. Bergman, E. Carpenter, R. Letelier, et al., 2002: Dinitrogen fixation in the world's oceans, *Biogeochemistry.* In press.

Leneman, H., G.W.J. Giessen, and P.B.M. Berentsen, 1993: Costs of reducing nitrogen and P emissions on pig farms in the Netherlands, *Journal of Environmental Management,* **39,** pp. 107–19.

Lyasse, O., B.K. Tosah, B. Vanlauwe, J. Diels, N. Sanginga, et al., 2002: Options for increasing P availability for low reactive phosphate rock. In: *Integrated Plant Nutrient Management in Sub-Saharan Africa: From Concept to Practice,* B. Vanlauwe, J. Diels, N. Sanginga, and R. Merckx (eds.), CAB International, Wallingford, UK, pp. 225–37.

Malik, A.S., D. Letson, and S.R. Chutchfield, 1993: Point/nonpoint sources trading of pollution abatement: Choosing the right trading ratio, *American Journal of Agriculture and Economics,* **75,** pp. 959–67.

Malik, A.S., B.A. Larson, and M.O. Ribaudo, 1994: Economic incentives for agricultural non-point source pollution control, *Water Resources Bulletin,* **30,** pp. 471–80.

Manyong, V.M., K.O. Makinde, and A.G.O. Ogungbile, 2002: Agricultural transformation and fertilizer use in the cereal-based systems of the Northern Guinea savannah, Nigeria. In: *Integrated Plant Nutrient Management in Sub-Saharan Africa,* B. Vanlauwe, J. Diels, N. Sanginga, and R. Merckx, (eds.), CAB International, Wallingford, UK, pp. 75–85.

Martinelli, L.A., 2003: Element interactions as influenced by human intervention. In: *Element Interactions: Rapid Assessment Project of SCOPE,* J.M. Melillo, C.B. Field, and B. Moldan (eds.), Island Press, Washington, DC.

Martinelli, L.A., A. Krusche, R.L. Victoria, P.B. Camargo, M.C. Bernardes, et al., 1999: Effects of sewage on the chemical composition of Piracicaba River, Brazil, *Water Air and Soil Pollution,* **110,** pp. 67–79.

McIsaac, G.F., M.B. David, G.Z. Gertner, and D.A. Goolsby, 2001: Eutrophication: Nitrate flux in the Mississippi River, *Nature,* **414,** pp. 166–67.

Merckx, R., 2002: Process research and soil fertility in Africa: Who cares? In: *Integrated Plant Nutrient Management in Sub-Saharan Africa: From Concept to Practice,* B. Vanlauwe, J. Diels, N. Sanginga, and R. Merckx (eds.), CAB International, Wallingford, UK, pp. 97–111.

Mitsch, W.J., J. Day, J.W. Gilliam, P.M. Groffman, D.L. Hey, G.W. Randall, and N. Wang, 1999: *Reducing Nutrient Loads, Especially Nitrate-Nitrogen, to Surface Water, Ground Water, and the Gulf of Mexico, Topic 5,* Report for the Integrated Assessment on Hypoxia in the Gulf of Mexico, Decision Analysis Series No. 19, NOAA Coastal Ocean Program, National Oceanic and Atmospheric Administration, Silver Spring, MD.

Mitsch, W.J., J. Day, J.W. Gilliam, P.M. Groffman, D.L. Hey, et al., 2001: Reducing nitrogen loading to the Gulf of Mexico from the Mississippi River basin: Strategies to counter a persistent ecological problem, *BioScience,* **51,** pp. 373–88.

Mokwunye, U. and A. Bationo, 2002: Meeting the P needs of the soils and crops of West Africa: The role of indigenous phosphate rocks. In: *Integrated Plant Nutrient Management in Sub-Saharan Africa: From Concept to Practice,* B. Vanlauwe, J. Diels, N. Sanginga, and R. Merckx (eds.), CAB International, Wallingford, UK, pp. 209–24.

Moomaw, W.R., 2002: Energy, industry, and nitrogen: Strategies for decreasing reactive nitrogen emissions, *Ambio,* **31(2),** pp. 184–99.

Mosier, A.R., M.A. Bleken, P. Chaiwanakupt, E.C. Ellis, J.R. Freney, et al., 2001: Policy implications of human-accelerated nitrogen cycling, *Biogeochemistry,* **52,** pp. 281–320.

Nixon, S.W., 1995: Coastal marine eutrophication: A definition, social causes, and future concerns, *Ophelia,* **41,** pp. 199–219.

Nixon, S.W., J.W. Ammerman, L.P. Atkinson, V.M. Berounsky, G. Billen, et al., 1996: The fate of nitrogen and P at the land-sea margin of the North Atlantic Ocean, *Biogeochemistry,* **35,** pp. 141–80.

NRC (National Research Council), 1993a: *Managing Wastewater in Coastal Urban Areas,* National Academy Press, Washington, DC.

NRC, 1993b: *Soil and Water Quality: An Agenda for Agriculture,* National Academy Press, Washington, DC.

NRC, 2000: *Clean Coastal Waters: Understanding and Reducing the Effects of Nutrient Pollution,* National Academy Press, Washington, DC.

NRC, 2001: *Assessing the Total Maximum Daily Load Approach to Water Quality Management,* National Academy Press, Washington, DC.

Oppenheim, S., 2001: Alternative agriculture in Cuba, *American Entomologist,* **47(4),** pp. 216–27.

OSPAR Commission, 2000: *Quality Status Report 2000,* OSPAR (Oslo-Paris) Commission, London, 108 + vii pp.

Paarlberg, R.L., 1996: Rice bowls and dust bowls: Africa, not China, faces a food crisis, *Foreign Affairs,* **75(3),** pp. 127–132.

Palm, C.A., R.J.K. Myers, and S.M. Nandwa, 1997: Combined use of organic and inorganic nutrient sources for soil fertility maintenance and replenishment. In: *Replenishing Soil Fertility in Africa,* R.J. Buresh, P.A. Sanchez, and F. Calhoun (eds.), SSSA special publication no. 51, Soil Science Society of America, Madison, WI, pp. 193–217.

Palm, C.A., K.E. Giller, P.L. Mafongoya, and M.J. Swift, 2001: Management of organic matter in the tropics: Translating theory into practice, *Nutrient Cycling in Agroecosystems,* **61,** 63–75.

Peterson, B.J., W.M. Wollheim, P.J. Mulholland, J.R. Webster, J.L. Meyer, et al., 2001: Control of nitrogen exports from watersheds by headwater streams, *Science,* **292,** 86 pp.

Powell, J., 2001: *Programs that Reduce Nutrient Pollution in Coastal Waters,* Unpublished report to the Pew Oceans Commission in Philadephia, PA, and Washington, DC.

Rabalais, N.N., 2002: Nitrogen in aquatic ecosystems, *Ambio,* **31,** pp. 102–12.

Randall, G.W. and D.J. Mulla, 2001: Nitrate nitrogen in surface waters as influence by climatic conditions and agricultural practices, *Journal of Environmental Quality,* **30,** pp. 337–44.

Randall, G.W., D.R. Huggins, M.P. Russelle, D.J. Fuchs, W.W. Nelson, et al., 1997: Nitrate losses through subsurface tile drainage in CRP, alfalfa, and row crop systems, *Journal of Environmental Quality,* **26,** pp. 1240–47.

Robinson, J.R. and T.L. Napier, 2002: Adoption of nutrient management techniques to reduce hypoxia in the Gulf of Mexico, *Agricultural Systems,* **72,** pp. 197–213.

Ryther, J.H. and W.M. Dunstan, 1971: Nitrogen, P, and eutrophication in the coastal marine environment, *Science,* **171,** pp. 1008–12.

Sahrawat, K.L., M.K. Abekoe, and S. Diatta, 2001: Application of inorganic P fertilizer. In: *Sustaining Soil Fertility in West Africa,* G. Tian, F. Ishida, and D. Keatinge, (eds.), SSSA special publication no. 58, Soil Science Society of America, Madison, Wisconsin, pp. 225–46.

Sanchez, P.A., K.D. Shepherd, M.J. Soule, F.M. Place, R.J. Buresh, et al., 1997: In: *Replenishing Soil Fertility in Africa,* R.J. Buresh, P.A. Sanchez, and F. Calhoun, (eds.), SSSA special publication no. 51, Soil Science Society of America, Madison, Wisconsin, pp. 1–46.

Sanchez, P.A., 2002: Soil fertility and hunger in Africa, *Science,* **295,** pp. 2019–20.

Scoones, I. and C. Toulmin, 1999: Policies for soil fertility management in Africa, A report prepared for the DFID (Department for International Development), IDS (Institute of Development Studies) Brighton/IIED (International Institute for Environment and Development), Edinburgh, UK, 128 pp.

Shuiwang, D., Z. Shen, and H. Hongyu, 2000: Transport of dissolved nitrogen from the major rivers to estuaries in China, *Nutrient Cycling in Agroecosystems,* **57,** pp. 13–22.

Shortle, J.S. and J.W. Dunn, 1986: The relative efficiency of agricultural source water pollution control policies, *American Journal of Agricultural Economics,* **68,** 668–77.

Smil, V., 2001: *Enriching the Earth,* MIT Press, Cambridge, MA.

Staver, K.W. and R.B. Brinsfield, 1998: Use of cereal grain winter cover crops to reduce groundwater nitrate contamination in the mid-Atlantic coastal plain, *Journal of Soil Water Conserv*ation, **53,** pp. 230–40.

Stoorvogel, J.J., and E.M.A. Smaling, 1990: *Assessment of Soil Nutrient Depletion in Sub-Saharan Africa: 1983–2000, Vol. I, Main report (2nd ed.), Report 28,* The Winand Staring Centre for Integrated Land, Soil, and Water Research, Wageningen, The Netherlands.

Tartowski, S. and R.W. Howarth, 2000: Nitrogen, nitrogen cycling*, Encyclopedia of Biodiversity,* **4,** pp. 377–88.

Townsend, A.R., R. Howarth, F.A. Bazzaz, M.S. Booth, C.C. Cleveland, et al., 2003: Human health effects of a changing global nitrogen cycle, *Frontiers in Ecology & Environment,* **1,** 240–46.

UNEP (United Nations Environment Programme), 2002: *Global Environment Outlook 3,* Earthscan Publications Ltd., London, UK.

USEPA (United States Environmental Protection Agency), 2000*: National Air Pollutant Emission Trends 1900–1998,* EPA-454/R-00–002, USEPA, Washington, DC.

USEPA, 2001a: *Nutrient Criteria Technical Guidance Manual for Estuarine and Coastal Marine Waters,* EPA-822-B-01–003, USEPA, Washington, DC.

USEPA, 2001b: *Market-based Approaches to Improve the Nations Waters,* Draft, 21 November 2001, USEPA, Office of Water, Washington, DC.

USEPA, 2003: Concentrated animal feeding operations (CAFO) – Final rule. Available at http://cfpub.epa.gov/npdes/afo/cafofinalrule.cfm, and http://www.epa.gov/npdes/pubs/cafo_brochure_regulated.pdf.

Van Asseldonk, M.A.P.M., 1994: Voorspellen van variatie tussen dieren met het Technisch Model Varkensvoeding, M.Sc. thesis, Department of Agricultural Economics, Wageningen Agricultural University, Wageningen, The Netherlands.

Vitousek, P.M., J.D. Aber, R.W. Howarth, G.E. Likens, P.A. Matson, et al., 1997: Human alteration of the global nitrogen cycle: Sources and consequences, *Ecological Applications,* **7,** pp. 737–50.

Vollenweider, R.A., 1968: Scientific fundamentals of the eutrophication of lakes and flowing waters, with particular reference to nitrogen and P as factors in eutrophication, Technical report DA 5/SCI/68.27, Organization for Economic Co-operation and Development, Paris, France, 250 pp.

Von Egmond, K., T. Bresser, and L. Bouwman, 2002: The European nitrogen case, *Ambio,* **31,** pp. 72–8.

Wolfe, A. H. and J.A. Patz, 2002: Reactive nitrogen and human health: Acute and long-term implications, *Ambio,* **31,** pp. 120–25.

Xing, G.X. and Z.L. Zhu, 2002: Regional nitrogen budgets for China and its major watersheds, *Biogeochemistry,* **57/58,** pp. 405–27.