Global Loss of Coastal Habitats
Rates, Causes and Consequences

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1. NUTRIENT POLLUTION, EUTROPHICATION, AND THE DEGRADATION OF COASTAL MARINE ECOSYSTEMS

by

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1.1. INTRODUCTION

IF A COASTAL MARINE ECOLOGIST had been asked a century ago what the most dangerous things that people put into the sea were, he would probably have settled on the various types of contagion that made people sick with typhoid, cholera, and dysentery. Floating filth, such as the remains of carcasses from slaughterhouses, might also have made his list. Fifty years ago the same question might have generated answers implicating oil, heavy metals, pesticides, and vast quantities of organic matter (largely from human sewage) that consumed much of the oxygen in tidal rivers and estuaries. Thanks to great advances in sanitary engineering, enhanced environmental consciousness and enormous investments in sewage treatment infrastructure in many parts of the world, today's marine ecologist would almost certainly have a very different set of things on her list. The three most dangerous things that we put into the sea today may well be fresh water, fishing nets, and nutrients.

While sea level rise from melting glaciers and overfishing from greed and inept management are clearly great threats to coastal marine ecosystems around the world, our purpose in this chapter is to focus on nutrients, especially nitrogen, and their link to eutrophication. Nutrient pollution is perhaps less widely recognized as a threat to coastal marine ecosystems than sea level rise or overfishing, but the issue began receiving a lot of political attention in much of northwestern Europe some thirty years ago (deJong 2006). There is continuing attention to the problem among coastal managers in the United States (e.g., Bricker et al. 2007), Europe (e.g., Ærtebjerg, Andersen, and Hansen 2003; Langmead and McQuatters-Gollop 2007), and internationally (e.g., UNEP and WHRC 2007; SCOPE 2007; Selman 2007; INI 2007).

◀ **Photo 1.1: Coral reefs are among the most nutrient sensitive coastal marine ecosystems.** This reef formation lies in the crystal clear waters of the Red Sea off Ras Mohammed, Egypt.

1.1.1. Some definitions

In spite of an effort to provide a simple operational definition of eutrophication over a decade ago (Nixon 1995), the term is still used in fuzzy and often confusing ways by scientists and managers alike. To some, the term means high concentrations of nutrients (usually nitrogen, N and/or phosphorus, P), or high inputs of nutrients, or low concentrations of dissolved oxygen, or high concentrations of chlorophyll, or large amounts of algae or dead fish on beaches, or foul smelling air. But eutrophication is actually much more interesting and important:

– *Eutrophication* (noun)—an increase in the rate of supply of organic matter to an ecosystem.

This definition emphasizes that eutrophication is a *process*, a change, an increase in the organic carbon (C) and energy available to the ecosystem—it is not a condition. Some confusion arises because ecologists use the term "eutrophic" to characterize systems that have high primary production (the rate of carbon fixation or formation of new organic matter from carbon dioxide and nutrients). All of the conditions listed above may be found in coastal marine ecosystems that are eutrophic, but they are not necessarily indicators of eutrophication. There is no universally accepted standard for the level of production that must be present for a marine ecosystem to be considered eutrophic. One frequently used guideline is 300 to 500 g C m^{-2} y^{-1} (Nixon 1995). There is a possibility that some marine waters may always have been eutrophic, including upwelling areas off the coast of Peru and parts of Africa. Many others have become eutrophic because of eutrophication brought on by human actions. For example, some parts of the Baltic may be undergoing eutrophication as their primary production rises from 20 to 40 g C m^{-2} y^{-1}, but they are not yet eutrophic. By the same token, an estuary with relatively stable average production of 350 g C m^{-2} y^{-1} may be eutrophic, but it is not experiencing eutrophication.

When defined as above, there are two types of marine eutrophication that are closely related but different in some important ways. Unfortunately, the terms ecologists use to refer to them are awkward:

– *Allocthonous eutrophication*—when the increasing supply of organic matter to the ecosystem comes from outside the system.

– *Autochthonous eutrophication*—when the increasing supply of organic matter comes from increasing primary production within the system.

1.1.2. Organic loading from sewage and industrial wastes

The first great wave of coastal marine eutrophication was allocthonous and occurred in urban coastal areas beginning in the second half of the nineteenth century as public water supplies and then sewer systems were installed in wealthier cities in Europe and North America (e.g., Tarr 1971, 1996; Wood 1982; Nixon 1995; Melosi 2000; Nixon et al. 2008). Large amounts of organic matter from some forms of industry (e.g., food processing, paper, textiles) and human sewage were collected and efficiently carried to rivers draining to the sea or discharged directly in bays and estuaries. Public health impacts, such as the consequences of drinking contaminated water and eating contaminated shell fish, and obvious aesthetic considerations quickly made it apparent that some form of treatment was needed. For the most part, this consisted of screening, settling, and chlorination in the primary treatment of sewage. While this was largely effective in protecting human health and sensibilities, it did little to reduce the organic loading to coastal waters, and oxygen conditions in many urban estuaries deteriorated dramatically. The low (hypoxic) and complete absence of dissolved oxygen (anoxic) conditions began to reduce the abundance and diversity of bottom animals, block anadromous fish migra-

Photo 1.2: Sewage effluent. The plumbing of cities to supply water for drinking and fire protection and to remove water from sewage, industrial waste, and storm water runoff made it easy to transfer nutrients from the land to coastal waters.

tions, produce fish kills, and stimulate the production of noxious hydrogen sulfide gas that occasionally blackened the lead-based paint on waterfront houses. In temperate areas, many of the ecological impacts of increasing the supply of organic matter from land to coastal waters were thoroughly studied and documented during the 1950s to 1970s (e.g., review by Cronin 1967; McIntyre 1977; Pearson and Rosenberg 1978; Warwick and Clarke 1994). In many cases, a dramatic reduction in organic loading to estuaries did not come until the environmental movement of the 1960s and 1970s brought full secondary sewage treatment to the cities of the developed nations. Secondary treatment reduces markedly the biological oxygen demand or BOD of sewage effluent. The untreated discharge of large amounts of organic matter in sewage remains a problem in many developing countries, even where primary chlorination protects human health.

1.1.3. Nutrient enrichment

Autochthonous eutrophication emerged as a serious concern in the coastal marine environment much more recently (Nixon 1995). By far the most common cause of this type of eutrophication is anthropogenic enrichment with the fertilizing nutrients N and P. In some ways it is surprising that these were not widely recognized as potentially important pollutants of coastal marine ecosystems until the late 1960s and 1970s (Wulff 1990; Nixon 1995 and in press; Howarth and Marino 2006). While limnologists were ahead of marine ecologists in recognizing the impact of nutrient enrichment (e.g., National Academy of Sciences 1969), the central role of P in lake eutrophication was also not fixed conclusively until the 1970s (reviewed by Schindler 2006).

Although nutrient enrichment is by far the most common cause of coastal marine and freshwater autochthonous eutrophication, it is useful to note that it is not the only cause. Other changes can also increase the supply of organic matter from primary production within a bay or estuary (e.g., Cloern 2001; Caraco, Cole, and Strayer 2006). For example, dams constructed in the watershed commonly reduce the transport of suspended sediment downstream to an estuary. This can increase the clarity of the water in a previously turbid estuary and thus increase primary production. If chemicals toxic to marine phytoplankton are removed by waste water treatment (for example copper by industrial pre-treatment), primary production might increase. Filling across the mouth of an estuary or lagoon for road construction might increase the water residence time in the system and thus increase production. Human (or other) predators might consume filter feeding shell fish or prey on zooplankton that graze on phytoplankton, and thus increase primary production. And large-scale changes in climate and/or hydrography may act to increase production in complex ways that are not yet fully understood: for example, the recent increases in the abundance of phytoplankton in the North Sea and northeast Atlantic (Richardson and Schoeman 2004; McQuatters-Gollop et al. 2007).

Such interesting exceptions aside, there is no question that anthropogenic nutrient enrichment is responsible for the vast majority of coastal ecosystems experiencing eutrophication, now or in future. And it is clear that nutrient-driven coastal eutrophication has been increasing dramatically in recent decades. Ivan Valiela summarized it well in his excellent new book on global coastal change (Valiela 2006): “Even within the limitations of available information, it was evident that [coastal marine] eutrophication was widespread, and increasing, into the 21st century.” Autochthonous eutrophication from nutrient fertilization is much more widespread and damaging than that caused by organic loading. It is not restricted to coastal waters surrounding large urban or industrial areas and, once added to an ecosystem, N and P can be recycled many times. In other words, the inorganic N or P added to the system stimulates the production of organic matter by plants. As this organic matter dies and decomposes, it consumes dissolved oxygen. However, the decomposition also releases the N and P which can then be used again by plants to fix yet more organic matter. This recycling may occur many times before an atom of N or P is flushed from an estuary.

Of course, the organic matter added to rivers and estuaries by sewage treatment plants also contained N and P, so the early allocthonous eutrophication also produced local autochthonous eutrophication. In reading the historical literature, it is clear that this complication was little appreciated by urban sanitarians or marine biologists—the much more dramatic and visible local impacts of massive organic loading largely overshadowed nutrient enrichment. If nutrient enrichment had been considered at all during the late 1800s and the first half of the 1900s, it would almost certainly been seen in a positive light as stimulating natural productivity along the coast (Johnstone 1908; Nixon and Buckley 2002; Nixon, in press).

The first implication of inorganic nutrients as an anthropogenic pollutant with negative impacts in the coastal marine environment appears to have been a result of the studies of phytoplankton blooms (“green tides”) conducted by John Ryther (1954, 1989) in Great South Bay and Moriches Bay on Long Island, New York. This work identified nitrogen enrichment from duck farms as the probable cause of the blooms and set the stage for a later paper that would have a much greater impact. The publication in 1971 of "Nitrogen, phosphorus, and eutrophication in the coastal marine environment" by Ryther and Dunstan in *Science* magazine clearly focused the attention of the marine research community on inorganic N as the nutrient whose supply most commonly limited the growth of phytoplankton in coastal waters. This set marine eutrophication apart from the more established paradigm of P limitation in lakes, and stimulated decades of research and management focused on N in coastal areas. In truth, however, the Ryther and Dunstan (1971) paper was the rediscovery of a view established seventy years earlier by the work of marine scientists in Europe. As Mills (1989) noted in his outstanding history of biological oceanography: "The history of [marine] plankton dynamics after 1899 is largely the history of the nitrogen cycle." While the role of N as the most common and pervasive limiting nutrient in temperate marine coastal waters has been confirmed repeatedly in bioassays, mesocosm experiments, numerical models, and stoichiometric analyses, it has also become clear that P limitation may be important in some parts of some estuaries, especially during times of high freshwater inflow (Howarth and Marino 2006). It is also clear that P limitation may be more common in tropical systems with carbonate sediments that can bind tightly with P (e.g., Nielsen, Koch, and Madden 2007). Because of the well recognized importance of N pollution in contributing to the eutrophication of most temperate (and many tropical) coastal ecosystems, most of this discussion will focus on N, including its sources, its pathways of entry to the coastal marine environment, and its effects. These are all topics that have received a great deal of attention in the scientific literature and in the popular press in recent decades. Scientific compilations include special issues of the journals *Estuaries* (Rabalais and Nixon 2002), *Ambio* (Galloway and Cowling 2002), *Limnology and Oceanography* (Smith, Joy, and Howarth 2006), and *Ecological Applications* (Kennish and Townsend 2007). Good non-technical overviews are given in two brief "white papers" from the Ecological Society of America (Vitousek et al. 1997 and Howarth et al. 2000), and in more extended form in *Global Coastal Change* (Valiela 2006).

1.2. NITROGEN AND EUTROPHICATION IN COASTAL MARINE SYSTEMS

Nitrogen pollution has a number of consequences in coastal marine ecosystems, in addition to stimulating an increase in the amount of organic matter being produced. Among some of the more thoroughly documented is chang-

Photo 1.3: Adult of the endangered green sea turtle (*Chelonia mydas*), a species which grazes on seagrass. These grasses do not survive in nutrient enriched waters, where they are shaded out by phytoplankton blooms.

ing the type and species of plants that make the organic matter. This may take the form of subtle shifts in the species composition of phytoplankton (e.g., Turner 2002) or more conspicuous changes in the types of plants supporting the ecosystem. Changes in the species and size composition of the phytoplankton can have important implications for the grazing animals in the water column and on the bottom that feed on them (e.g., Olsen et al. 2006; Wolowicz et al. 2006). It is also possible that nutrient enrichment and eutrophication are contributing to the reported increases in harmful algal blooms around the world, but this linkage remains more controversial. As concluded by Anderson et al. (2002) after an extensive review, "… the relationships between nutrient delivery and the development of blooms and their potential toxicity or harmfulness remains poorly understood … Nutrient enrichment has been strongly linked to stimulation of some harmful species, but for others it has not been an apparent contributing factor."

It has become increasingly clear that N fertilization of shallow low nutrient waters where rooted seagrasses dominate can increase the fouling of the seagrass leaves by epiphytes, produce dense floating mats of drift macroalgae, and ultimately result in intense blooms of phytoplankton. All of these conspire to shade the seagrass to such an extent that it may be completely eliminated even at very low levels of nutrient enrichment (e.g., Twilley et al. 1985; Duarte 1995; Corredor et al. 1999; Nixon et al. 2001; Valiela 2006). There is also some experimental evidence from mesocosms that the impact of nitrogen on temperate coastal lagoons with eelgrass (*Zostera marina*) is exacerbated by even small increases in temperature (Bintz et al. 2003). Studies by Deegan (2002) have also shown that the habitat value of seagrass beds for fish may be seriously reduced by nutrient enrichment, well before the grasses are completely eliminated.

Coral reefs appear to be even more sensitive to nutrient enrichment than seagrass meadows (D'Elia 1988) and have been described as "… the most nutrient-sensitive of all ecosystems." (Goreau 2003). Perhaps the best documented demonstration of the impacts of nutrient enrichment on coral reefs comes from the detailed study of reef recovery in Kaneohe Bay, Hawaii following the diversion of sewage effluents (Smith et al. 1981; Nixon et al. 1986). Unfortunately, continued population growth in the Kaneohe Bay watershed and in non-point sources of N to the system appear to have reversed some of the recovery, and macroalgal overgrowth is once again a problem on the reefs (e.g., Stimson, Larned, and McDermid 1996). Coral reefs represent a case in which nutrient enrichment may cause dramatic species changes, habitat structural changes, and increased organic production simultaneously, as soft or fleshy macroalgae overgrow hard encrusting algae and coral. However, given the high complexity and great diversity of coral reefs, it is perhaps not surprising that the role of nutrient enrichment in coral reef degradation remains controversial within the scientific community (e.g., Lapointe 1997; Hughes et al. 1999; and Lapointe 1999). A recent review concluded that evidence for nutrient enrichment being a major cause of the world-wide degradation of coral reefs was "… equivocal at best." (Szmant 2002). The situation is complicated by the common co-occurrence of overfishing and nutrient enrichment, and some investigators have argued that the overharvesting of herbivorous fish and/or the loss of grazers (e.g., sea urchins) to disease have been more important than anthropogenic nutrient fertilization in promoting macroalgal overgrowth (Szmant 2002). In fact, a recent review has argued that many of the negative changes attributed to nutrient enrichment in seagrass, rocky intertidal, and coral reef communities are really due to human alterations of coastal food webs (Heck and Valentine 2007). On the other hand, several of the major studies supporting the importance of "top-down" or grazing effects on macroalgae on reefs have been vigorously criticized (Goreau 2003), and it seems compelling that nutrient enrichment can play an important role in local reef degradation. On a larger scale, storm damage, coral diseases, warming, and sedimentation must also be important factors (Rogers and Miller 2006).

Regardless of their obvious importance, these various responses to nitrogen enrichment are not, in themselves, eutrophication (with the possible exception of increases in net ecosystem production due to macroalgal growth on coral reefs). They are responses to nutrient enrichment, certainly, but they may or may not be associated with an increase in the production of organic matter in the system. When eutrophication does occur, it may be associated with these or other changes, some of which may be seen as desirable and others not. Among the desirable changes in phytoplankton-based systems may be an increase in benthic animals and the production of harvestable fish, at least up to some point at which hypoxia or anoxia may outweigh the positive influence of a greater food supply (Nixon 1988; Caddy 1993; Herman et al. 1999; Breitburg 2002; Nixon and Buckley 2002; Kemp et al. 2005; Oczkowski and Nixon 2008). And it is the occurrence of hypoxia and anoxia that is the best documented and understood and, perhaps, most severe impact of eutrophication (e.g., Diaz and Rosenberg 2001; Rabalais and Turner 2001). It is the link between N (or, in some cases, P) inputs and accelerated organic production and resulting low oxygen that is the most common concern for managers and marine ecologists. It is this threat that unifies allocthonous and autochthonous eutrophication and thus makes much of the research from earlier decades a helpful platform for understanding what may be the most widespread impact of nutrient pollution.

1.2.1. The oxygen problem

If you are not a limnologist or an oceanographer, you may find yourself puzzled by why we worry about fertilizing lakes and bays with nutrients and making the plants grow faster. And why more plants may mean less oxygen. After all, farmers and gardeners use nutrients to accelerate plant growth all the time on land. And there are popular bumper stickers asking if one has thanked a green plant lately—presumably for making oxygen for us to breathe. The reasons have to do with important differences between air and water. First, a cubic meter of air contains about 270 g of oxygen, while the same volume of sea water in equilibrium with the air only holds 5-10 g of oxygen, depending on its salinity and temperature (warmer and/or saltier holds less oxygen). But much more important is the fact that it takes very little energy to mix air—no one worries about having to keep moving to avoid consuming all the oxygen in the air in front of their face! Water is more viscous and it requires much more mechanical energy to provide turbulent mixing in water than in air. As a result it is quite possible for local oxygen to become depleted when winds or currents are not active. This is taken to an extreme when aquatic systems become vertically stratified in response to solar warming and/or freshwater inflows. Since estuaries are by definition semi-enclosed places where the salinity is diluted by fresh water (Pritchard 1967), they are susceptible to both agents of stratification. Solar energy warms the surface waters and thus makes them less dense than the cooler water below. Fresh water is less dense than salty water and tends to float on the surface. The greater the density difference between the warmer fresher surface water and the cooler saltier bottom water, the more wind and tidal energy is needed to mix them. When the water is strongly stratified, the deeper water may not come into contact with the air for many days or even months. As respiration of organisms in the deeper water and in the bottom sediments proceeds, especially at the higher rates that come with higher summer temperatures, the oxygen in the bottom water becomes more and more depleted. Once it is completely consumed and the water and sediments are anoxic, toxic hydrogen sulfide is produced. In this way even some organisms that can tolerate low or even no oxygen conditions for short times may be killed. While mobile animals like fish can usually avoid hypoxic and anoxic areas, they sometimes become trapped against the shore and cannot escape. In some other situations, wind and tidal mixing may be so weak and respiration rates so high that even the surface waters can become hypoxic or (rarely) anoxic and cause fish kills.

Conspicuous blooms of macroalgae and phytoplankton that may result from nutrient enrichment do produce oxygen as land plants do, but this takes place only during the day when the plants are actively growing. The surface waters where light is plentiful may even become supersaturated with oxygen, which diffuses out into the air. At night, when there is no oxygen production but lots of respiration, the "lost" oxygen made during the day when the plants were growing is no longer available, and oxygen levels may become very low if respiration demands exceed the rate at which oxygen can diffuse back into the water from the air. Even more problematic is the fact that the macroalgae and phytoplankton do not stay in the surface water where they grow. They sink into the deeper water as they die, or are eaten by grazing animals and excreted as fecal pellets. In stratified systems, this rain of organic matter stimulates respiration in the isolated bottom water and sediments, which depletes bottom water oxygen levels.

While it appears that the number of coastal areas experiencing hypoxia and anoxia is increasing, especially in Europe and North America, and that the aerial and temporal extent and intensity of hypoxia is increasing (Diaz 2001; Selman 2007), it must be remembered that oxygen concentrations vary a great deal in many coastal systems from day to day and, in fact, from hour to hour with light and tides. They also vary strongly in many areas with depth and with the history of wind and tidal mixing. It is also true that as the research and management communities became more aware of the nutrient-eutrophication-hypoxia/anoxia linkage, they focused more efforts on measuring dissolved oxygen. And advances in instrumentation have made it increasingly practical to deploy oxygen meters for continuous recording of dissolved oxygen over long periods of time. For hypoxia, as for many other things, the more you look, the more you find. On the other hand, it is also easy to miss hypoxic conditions—bottom waters that have experienced low oxygen for days may recover within minutes or hours with a strong wind. Hypoxia is a dark shadow that is difficult to scale and track precisely. But surveys of scientific opinion in the U.S. and Europe clearly show widespread concern about eutrophication and hypoxia (Bricker et al. 2007; Langmead and McQuatters-Gollop 2007; Selman 2007), and there is no reason to doubt that warming waters that are receiving ever more N and P are likely to be experiencing increasing hypoxia and anoxia. As Valiela (2006) put it: "It seems safe to conclude that most coastal waters are exposed to some degree of eutrophication, and that in most of these cases conditions are worsening."

1.3. WHY NITROGEN IS DIFFICULT TO CONTROL

1.3.1. Sources are irreplaceable, complex, and widespread

Anthropogenic N enters the coastal marine environment because of two essential human activities—the combustion of organic matter to release energy (including biomass, coal, oil, and natural gas) and the production and consumption of food (Galloway et al. 2002). In the case of coal combustion (and to a lesser extent crude oil combustion), some fossil N is released from the fuel itself, and some is "fixed" or made available to most plants by the oxidation of N in the atmosphere at high temperatures. Biomass burning releases N contained in the organic matter and fixes N from the atmosphere. The combustion of natural gas only fixes N from the atmosphere. Since N accounts for almost 80% of the atmosphere, the potential supply of N from this source is inexhaustible (e.g., Galloway et al. 2002). Because the release and production

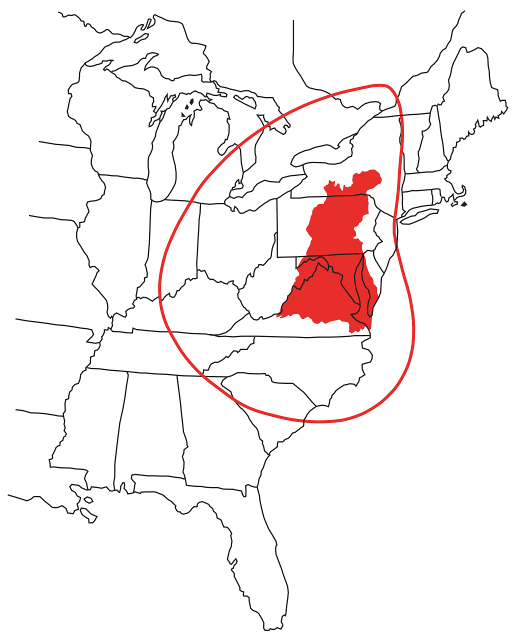
Photo 1.4: A portion of the shoreline of Chesapeake Bay, the largest estuary in the United States. The inputs of nitrogen to a system like this come from many different sources and are difficult to control.

of reactive N is an inadvertent consequence of fuel combustion, N pollution and the problem of increasing atmospheric CO_2 are linked, though the choice of fuel and improving technology can change the link in important ways (Galloway and Cowling 2002). Because the oxidized atmospheric N appears as nitric acid in rain, N pollution and lake and forest acidification are also linked. Because fuel combustion puts reactive or biologically available N into the atmosphere, that N can easily travel great distances before it is deposited on land and water. This means that N can be deposited on coastal watersheds and coastal marine waters from sources far from the coast and outside of the watershed draining to a bay or estuary. The area from which various materials may be put into the atmosphere and reach a given estuary is called the airshed of that estuary. Because different materials behave differently in the atmosphere, the boundaries of the airshed vary for different pollutants. As an example, N modeling studies suggest that the airshed of Chesapeake Bay is 6.5 times larger than the watershed of the bay, which is itself 17 times bigger than the bay (Chesapeake Bay Program undated) (figure 1.1).

Combustion sources of reactive N are both fixed (e.g., electric power generation plants, industries) and mobile (e.g., road and air transport). The importance of various sources varies around the world. For example, road transport accounted for about 28% of N oxide emissions in Asia in 1990 but for 45% of emissions in Europe in 1998 (Bradley and Jones 2002). Electric power generation contributes a larger share of N oxide emissions in coal burning Asia (~ 31%) than it does in Europe and North America, which rely more on oil, natural gas, and nuclear energy for electric power generation (Bradley and Jones 2002).

Not surprisingly, the global distribution of the deposition of reactive N from the atmosphere corresponds closely to the global distribution of fossil fuel combustion (and human population density) (e.g., Galloway and Cowling

Figure 1.1: Airshed and watershed of Chesapeake Bay. The area of the airshed is over six times as great as the area of the Chesapeake Bay watershed.



Source: http://www.epa.gov/AMD/images/chesbay.oxN.gif.

2002). It is more difficult to assess the amount of N arriving from atmospheric deposition that actually enters a particular coastal water body. Some is deposited directly on the water surface, and the relative importance of this input compared to inputs from the watershed or catchment tends to vary directly with the size of the water body (e.g., Paerl 1995). However, some fraction of the N that is transported through the atmosphere and deposited on the larger watershed will also ultimately reach downstream coastal waters. This may be more important than the direct deposition and is much more difficult to quantify. It is usually estimated using indirect modeling techniques or, more rarely, measurements of stable N isotopes in rivers (e.g., Howarth 1998; Mayer et al. 2002; Boyer et al. 2002).

Food production makes the N in the atmosphere available to the biosphere in two ways: from the industrial production of inorganic N fertilizers in the Haber-Bosh process; and from the cultivation of specialized N-fixing crops such as soybeans and pulses (Smil 2002). The combined production of reactive N in agriculture is over five times greater than that associated with fuel combustion (about 100 Tg N y^{-1} in Haber-Bosch, over 30 Tg y^{-1} in biological fixation, and about 25 Tg y^{-1} from combustion; Galloway et al. 2002). The most recent assessment of the global N budget suggests that total anthropogenic sources of N may now be about 1.7 times the estimated background sources, due to lightning and natural terrestrial and marine N fixation. This represents a very large perturbation of one of the biosphere's most important biogeochemical cycles.

As with fuel combustion, the production of synthetic fertilizer increased rapidly with economic expansion following the Second World War (Smil 2002) as part of what has been called "The Great Acceleration" (Steffen, Crutzen, and McNeill 2007). The absolute importance of synthetic N fertilizer to the current human population has been emphasized by Smil (2001) after extensive analysis:

– We can thus conclude that the Haber-Bosch synthesis now provides the very means of survival for about 40% of humanity; …

Our ever-increasing use of synthetic fertilizers has been driven by two important factors: increasing human population and a growing world economy (Steffen, Crutzen, and McNeill 2007). While the role of the first is obvious, the second may be less appreciated. There is a correlation between wealth among countries and their use of synthetic fertilizer (e.g., Nixon 1995). Much of this correlation may be due to another correlation, that between income and per capita protein consumption (Nixon 1995). The latter is important because it is the consumption of protein that provides N in the diet—N that is (except in growing children) ultimately excreted into the environment. Still more important, however, is the link between income and the type of protein consumed: vegetable protein or meat protein. While there are important cultural factors that influence the consumption of meat and the forms of meat consumed, the general pattern is that meat consumption increases markedly with growing wealth. This is shown very dramatically by an analysis of changing per capita gross domestic product and per capita consumption of meat, milk, eggs, and rice in thirteen Asian countries (Shindo, Okamoto, and Kawashima 2006). While the first three rose strongly with income, rice consumption showed little change or declined sharply as in South Korea and Malaysia. Even in a rich country like the United States, meat consumption has been rising steadily (Howarth et al. 2002).

The great range in per capita meat consumption and in the type of meat consumed is clearly evident in even a summary comparison of recent data for various countries (table 1.1). The U.S. mean of 126 kg per person per year is equivalent to 345 g per person per day. Since fresh meat of various types is about 20% protein (e.g., Held 2007), this converts to almost 70 g protein per person per day compared to recommended total (including plant protein) dietary intakes of 50 g per day for women over age 25 and 63 g per day for

Photo 1.5: Concentrated animal feeding operation. Modern beef production causes the direct and indirect addition of large amounts of nitrogen to the landscape.

Table 1.1: Annual meat consumption in various countries. Units are kg per person in 1999

Country	Total	Beef	Pork
United States	126	45	32
Denmark	114	21	74
Spain	104	16	64
France	89	27	38
Portugal	74	15	31
United Kingdom	73	20	25
Mexico	53	21	10
China	45	4	31
Ukraine	32	13	14
Egypt	16	8	–
India	2	1.5	–

Source: U.S. Census 2000. Per capita consumption of meat and poultry, by country statistics. Available at: http://www.allcountries.org/uscensus/1370_per_capita_consumption_of_meat_abd_html.

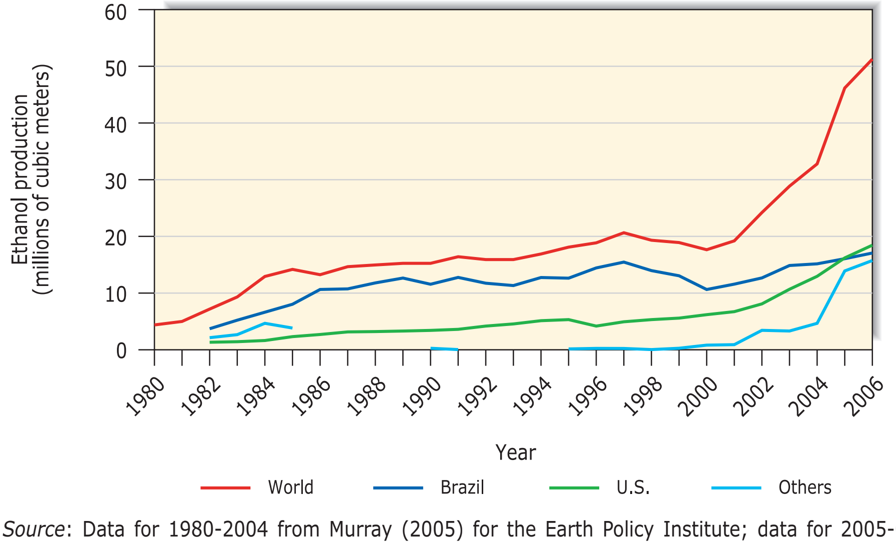
men over age 25 (National Academy of Sciences 1989). Of course, the population is not all over 25 years old, so the over consumption of protein is even greater than it appears. We are able to make a more detailed comparison of required vs. observed protein consumption for the city of Providence, Rhode Island (United States), where we have obtained extensive analyses of the N content of raw sewage entering the largest sewage treatment plant serving the city. These analyses suggest that the 100,000 plus population being served by the plant is consuming an average of just over 100 g of protein per person per day. This compares to an age (weight)- and gender-adjusted average recommended daily intake for the population of 50 g protein per person per day (Nixon et al. 2008). In other words, protein consumption in the city is roughly twice that needed for adequate nutrition, and twice as much N is being released in sewage as is required for the nutritional needs of the population. So high is the consumption of meat protein in the U.S. that a reduction of 40% would still leave the population with a per capita meat consumption equal to that of Great Britain; not a country known for vegetarianism!

The consumption of meat protein is of particular concern in terms of N pollution, not just because of the N excreted by meat-eating people. The production of meat is very inefficient in terms of N. In the United States, protein conversion efficiencies for edible portions of beef, pork, and chicken average about 5%, 10%, and 20%, respectively (Smil 2001). In other words, it requires 100 kg of N in corn (maize) to produce 5 kg of edible N in beef when averaged across the herd. The remaining 95 kg of N ultimately enters the landscape as metabolic waste from the cows or carcasses. In the last twenty years, the amount of grain being fed to animals has increased by 200-250 million tons (*The Economist* 2007). Moreover, the production of corn and other grains is not completely efficient in terms of N. Even in very efficient corn production in the U.S., recent N efficiency has been about 75%, meaning that about one quarter of the N applied as fertilizer does not enter the meat production food chain (Fixen and West 2002).

If this report had been written just a few years ago, our discussion of food production and coastal N pollution and eutrophication would have ended with the preceding paragraph. Today, however, we cannot leave this topic without noting the increasing link between what has been considered food production and fuel combustion.

This link arises because of the growing use of biomass (primarily sugar cane and corn or maize) to produce ethanol for use as an independent fuel or as a gasoline supplement in transportation. While this has been going on for over 25 years, the production of ethanol has increased dramatically in the last five years, especially in the United States (figure 1.2). Four countries, the U.S., Brazil, China, and India, now account for over 80% of global ethanol production (Murray 2005). While the combustion of ethanol in automobile engines oxidizes N from the atmosphere and makes it biologically available (as does the burning of gasoline), a major concern for marine ecologists is that both

Figure 1.2: Ethanol production throughout the world over the last twenty-five years



Source: Data for 1980-2004 from Murray (2005) for the Earth Policy Institute; data for 2005-06 from Renewable Fuels Association.

sugar cane (the major crop used in Brazil and other tropical countries) and maize (used in the U.S.) are crops that require large quantities of N fertilizer. Application of N in sugar cane production is commonly between 100 and 400 kg ha^{-1} y^{-1} (UN Food and Agricultural Organization online data, http://www.fao.org/docrep/007) and the average N application for U.S. corn is about 150 kg ha^{-1} y^{-1} (Fixen and West 2002). The rapid expansion of maize agriculture in the U.S. has come largely from the conversion of land formerly used for soybean and wheat production (*The Economist* 2007); crops requiring much less N fertilizer. Because soybeans grow in association with N-fixing bacteria, they may need relatively little or no synthetic N fertilizer (e.g., Staton and Warncke 2007), and N applications on wheat commonly range from about 50 to 75 kg ha^{-1} y^{-1} (Blumenthal and Sander 2002). It is worth asking how such land use change will impact the long-term plan to reduce nutrient loads to the Gulf of Mexico, and thus reduce the extent and severity of hypoxia in the northern Gulf (e.g., Rabalais et al. 2007; Justic et al. 2007).

The melding of the food and fuel economies is having dramatic impacts on the global price of food and on the ability of the U.S. to supply food to other countries. As noted in a recent essay, "The End of Cheap Food" (*The Economist* 2007): "The 30 m tonnes of extra maize going into ethanol this year amounts to half the fall in the world's overall grain stocks … : *fill up an SUV's fuel tank with ethanol and you have used enough maize to feed a person for a year.*"(emphasis added). While others have emphasized the questionable net energy yield of ethanol from maize, the impact of expanding sugar cane production on tropical forests, and the risks to global food security (e.g., Murray 2005), we believe that the rise of biomass-based ethanol production also poses risks for coastal marine ecosystems, especially the nutrient-sensitive tropical ones that lie downstream from sugar cane and other rapidly growing tropical plants.

1.3.2. Nitrogen moves in many forms and many ways

Throughout this report we have been referring to nitrogen as N, its symbol in the periodic table of elements. But N exists in many forms, and this chemical diversity complicates the measurement and management of the element as it moves from sources on land or in the atmosphere to the coast. The vast amount of N in the atmosphere exists as relatively inert N_2 gas that is only available to certain microbes with the special ability to "fix" or convert it into forms that are useable by other forms of life. Some of these N-fixing microbes live in close association with terrestrial plants, such as soybeans and alder trees, and can provide N for their needs. Some live in marine systems like coral reefs, salt marshes, and seagrass meadows or in surface waters of systems such as the Baltic Sea, and can add reactive N directly to the marine environment. Trace amounts of N also exist as N_2O, or nitrous oxide, a powerful greenhouse gas.

Reactive nitrogen produced by fuel combustion exists as various oxides of N or as ammonia and is in the form of gases, aerosols, and very fine particulates. There is also a significant amount of dissolved organic N in atmospheric deposition whose source(s) and fate is not well known. The transport and deposition of the different forms varies with temperature, the nature of the surface, and several other factors.

The production of synthetic fertilizer begins with the conversion of atmospheric N_2 into ammonium, but this can be converted into nitrate or urea and applied to fields in various ways. The amount of N that is lost from farm fields varies with the form of N applied, the method of application, the time of application, the type of crop being fertilized, and a number of other factors. Some of the N may be denitrified by special bacteria and returned to the atmosphere as N_2 gas. If animal manure is recycled as a source of organic N, much of that N may be lost to the atmosphere as ammonia gas that can be transported various distances before being redeposited, perhaps in coastal watersheds or directly in coastal waters. While some ammonium is absorbed in soils, some is oxidized to nitrite and nitrate, which can move easily through the soil in groundwater. Soil microbes also release dissolved organic N, a complex mix of poorly defined compounds from vegetation that may be easily taken up by other microbes or, in the case of some compounds, be very resistant to further biological activity. Much of the dissolved organic N also moves with groundwater and surface water to reach the coast, where its fate and impact are poorly known.

Nitrogen in the protein consumed by humans and other animals can reach the coastal marine environment by a variety of pathways. The N in animal waste can be deposited directly into streams, can be washed off impervious surfaces in concentrated animal feeding operations by wash water and storm water runoff, can be volatilized into the atmosphere, or enter groundwater. The N in human waste deposited in septic systems generally enters the groundwater, unless systems are specially designed for its removal. Similarly, the N in human waste that is collected by sewer systems can be transported even more efficiently into surface waters from sewage collection and, in some cases, treatment facilities. Because of its many sources and pathways, and because the airshed and watershed of most estuaries are much bigger than the estuary, N fertilization per area of estuaries is remarkably high: higher than the direct N fertilization of many major crops (table 1.2). Fortunately, advanced waste water treatment can be used to remove large amounts of the N in sewage, especially during warmer weather, though not without significant costs. Removing N from so-called "non-point sources" like agricultural runoff is much more challenging. In spite of all these complexities, an overall picture of the N links between airsheds, watersheds, and coastal ecosystems has emerged during the last decade or so. Surprisingly strong linear correlations have been found between the total input of anthropogenic N to watersheds (expressed per unit area) and the annual export of total N and dissolved inorganic N from the watershed (e.g. Peierls et al. 1991; Howarth 1998; Boyer et al. 2002). While the slope of the relationship appears to vary with temperature, such that warmer areas export a smaller fraction of the input (Schaefer and Alber 2007), the striking feature of the relationships is that relatively little of the N input leaves the watershed (Van Breemen 2002). Export in the northeastern U.S. averages about 25% of N input, against less than 10% in the southeastern U.S. (Schaefer and Alber 2007). The generality of these findings, particularly with respect to tropical watersheds with their strong wet-dry seasons, still needs to be determined, but they contain some good and some bad news for those concerned with coastal marine eutrophication. The good news is that watersheds with widely varying land use attenuate large amounts of N by sequestration and denitrification, and that warmer watersheds may be stronger sinks for N than we previously thought. The latter may be particularly important given the projected trends in tropical coastal areas discussed below. The bad news is that as more anthropogenic N enters a watershed, more N will reach the coast. It is also disquieting that such a large amount of N is retained and/or removed by processes that could be impacted by changing climate, potentially releasing previously stored N.

Table 1.2: N fertilization of agricultural crops and estuaries

Crops [1]	N, kg ha^{-1} y^{-1}	Estuaries [2]	N, kg ha^{-1} y^{-1}
Pineapple	500-650	Randers Fjord	2315
Bananas	300-600	Scheldt	1875
Rice	200-400	Lagoon of Venice	335
Potato	140-240	Narragansett Bay	275
Sugar cane	100-400	Chesapeake Bay	130
Corn (maize)	100-200	Baltic Sea	30
Spinach	60-100		

[1] *Source*: UN Food and Agricultural Organization, Department of Natural Resources Management and Environment: http://www.fao.org/documents and U.S. Department of Agriculture: http://www.ers.usda.gov/Data/FertilizerUse/.
[2] *Source*: Nixon and Pilson (1983).

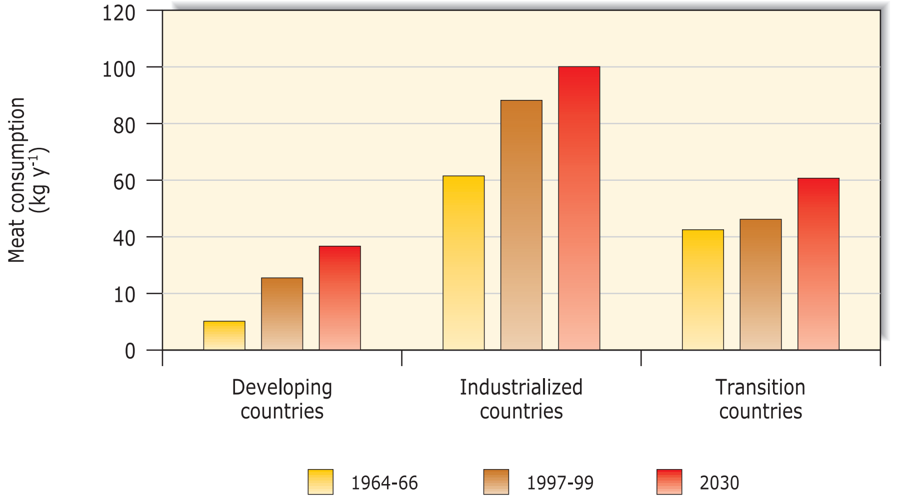
1.4. SOME CONCERNS FOR THE FUTURE

A detailed effort has recently been made to compare historical, current, and future (2050) global and regional N budgets (Galloway et al. 2002). The result suggests a future with much more reactive N moving through the biosphere, perhaps 70% more than under recent conditions. The model used in this study suggests a more modest increase of about 30% in the reactive N reaching the coast in rivers, though the authors caution that the model assumed that current rates of N attenuation in watersheds remain unchanged. They point out that this assumption may fail, as wetlands (important sites of N removal in watersheds) are increasingly filled, and as N deposition from the atmosphere increases markedly with increasing fossil fuel combustion. Atmospheric deposition may become an increasingly important pathway by which N reaches coastal ecosystems, unless the investment is made in improved technology to uncouple N emissions from combustion.

Future N pollution and coastal marine eutrophication will vary greatly in different parts of the world, with the greatest increases in Asia. As in the past, N pollution will follow economic expansion and population growth. As pointed out by Crutzen (2002), almost all the symptoms of "The Great Acceleration" have so far been caused by just 25% of the world population. As hundreds of millions of people in the developing world rapidly strive to attain Western standards of living, there will almost certainly be many surprises that even our most sophisticated models cannot foresee. For example, recent projections of N fertilizer use in the U.S. showed that diet choices could have a very significant impact (Howarth et al. 2002), but this exercise took place just before the food-for-fuel folly hit American agriculture. And at this writing there is no evidence of Americans reducing their consumption of meat, despite major education efforts by health agencies and the insurance industry to reduce the consumption of animal fat, and the growing awareness that we confront a national epidemic of obesity. One can only assume that the developing nations will continue to consume increasing amounts of meat (figure 1.3). Demand for livestock products has been growing three times faster in developing countries than in the industrialized world (Holmes 2001).

The developing world will also be the place where human population growth is greatest, and the most rapid growth will be in urban areas, most of which are on or near the coast (Laurence 2007). Urban growth is particularly important because it will require public health infrastructure in the form of water supply and sewage collection/disposal (Nixon 1995). Bush toilets and trenches may suffice in the country, but not in cities. As in Europe and North America in the late 1800s, this will bring increasing amounts of N and P to the coast (Nixon et al. 2008).

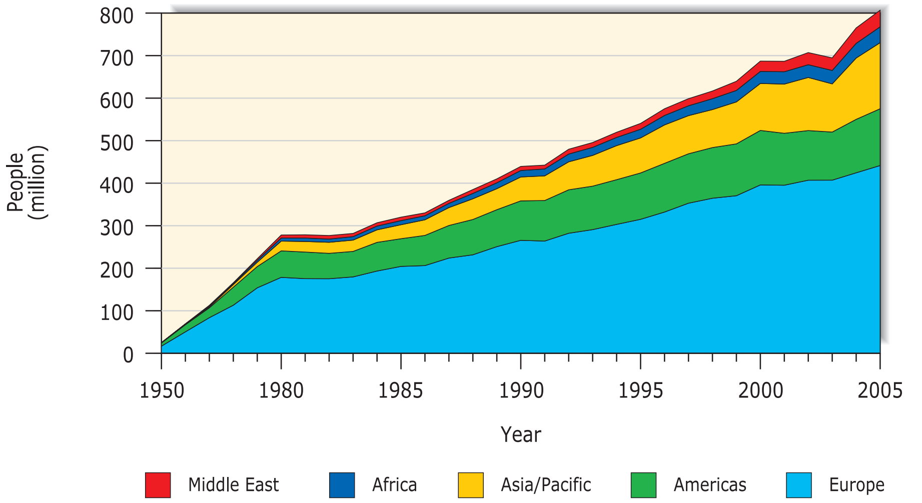
Figure 1.3: Per capita meat consumption. Consumption figures over the last forty years and projected into the future for developing, industrialized, and transition countries.



Source: WHO 2002. ftp://ftp.fao.org/docrep/fao/005/ac911e/ac911e00.pdf.

Most of the developing world lies in the tropics or subtropics, and it is the coastal marine ecosystems of these regions that will be most severely impacted by nutrient pollution in the coming decades. Many will be enriched by fertilizer runoff and livestock waste, some will be downstream of spreading aquaculture enterprises (also sources of N and P from fish or shrimp food and waste), almost all will be enriched by atmospheric N deposition from rapidly growing automobile fleets and increasing electric power generation, and some will receive increasing amounts of N and P from human sewage. Some tropical systems, especially coral reefs and seagrass meadows, may also be endangered by intensive development for coastal tourism. Globally, tourism accounts for approximately 35% of the world's exports of services and more than 70% in least developed countries (World Tourism Organization 2007). International tourism has also been part of "The Great Acceleration", increasing from fewer than 25 million travelers in 1950 to over 800 million in 2005 (figure 1.4). The most rapid increase has been in Asia and the Pacific at about 13% per year (World Tourism Organization 2007). Of course, not all tourism impacts the coastal environment, but the popularity of tropical beaches and coral reefs has certainly been growing. According to a recent assessment, 40% of the world's reefs are at risk from overexploitation, 30% are at risk from development, 20% suffer from inland pollution and erosion, and 10% are exposed to marine pollution (Bryant et al. 1998). Remarkably, just six coun-

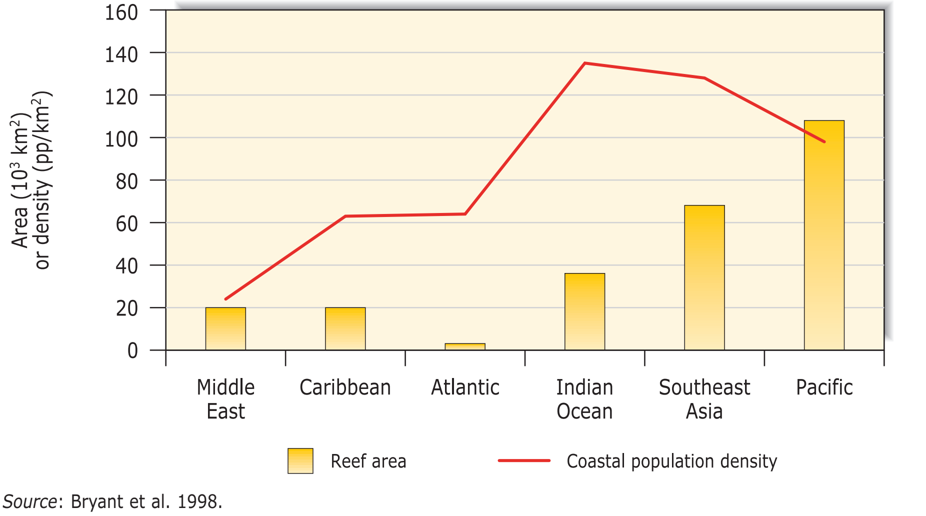
Figure 1.4: International tourist arrivals between 1950 and 2005



Source: WTO 2007. http://unwto.org/facts/eng/historical.htm.

tries contain over half the world's reefs: Australia, Indonesia, Philippines, Papua New Guinea, Fiji, and the Maldives. Reefs in Southeast Asia are the most threatened, with over 80% of them at risk, mainly from coastal development and overfishing (Bryant et al. 1998). Regions with high population density often have the most reef area (figure 1.5). Not surprisingly, there is a

Figure 1.5: Reef area by region and coastal population density



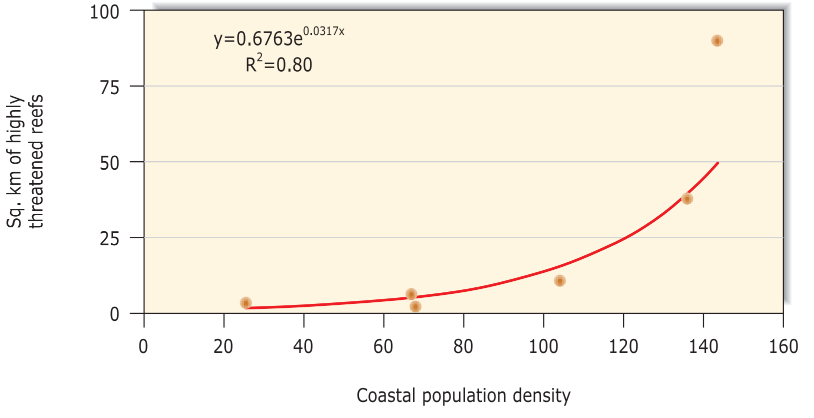
Source: Bryant et al. 1998.

Photo 1.6: Residential tourism development in a wetland area. Tropical coastal ecosystems will almost certainly experience increases in nutrient enrichment from growing tourism development, expanding agriculture, and rising coastal urban populations.

strong correlation between coastal population density and area of highly threatened reef (figure 1.6). While resort developers have probably learned the hard way not to let sewage contaminate the surrounding waters with pathogens, the threats posed by nutrient pollution have largely gone unnoticed (e.g., Goreau 2003). On the positive side, well-designed biological N removal in packaged sewage treatment plants constructed in association with resort development may be particularly effective at warm tropical temperatures.

In parts of Europe, North America, and other wealthy areas, the future of N pollution may be quite different, at least in urban estuaries where human sewage is the primary source of N. The rising awareness of problems associated with nutrient pollution during recent decades has led to increasing investment in advanced waste water treatment with N removal. Improved secondary treatment and removal of P from detergents has also led to declines in P loading (e.g., Nixon et al. 2008). Even in some areas with intensive agriculture, aggressive efforts to improve fertilizer efficiency and manure management have led to reduced nutrient pollution from these sources. For example, in Denmark: "Since 1991 land-based inputs of nitrogen and phosphorus to estu-

Figure 1.6: Reef area considered to be gravely threatened as a function of coastal population density



Source: Bryant et al. 1998.

aries and coastal areas have been reduced by 35% and 60% respectively. The reduction in nitrogen (21%) is mainly caused by reduced losses from agricultural soils, while the reduction in phosphorus is due to extension of sewage treatment."(Ærtebjerg, Andersen, and Hansen 2003, p. 107). These reductions in loading led (after a lag) to "significant decreases in nutrient concentrations on a large regional scale …", including the open waters of the Kattegat, the Sound, and the Belt Sea, as well as estuaries (Carstensen et al. 2006, p. 398). Primary production in these same areas increased from the 1950s through the 1980s, then declined modestly coincident with declining nutrient loads through the 1990s (Rydberg, Ærtebjerg, and Edler 2006). Unfortunately, in the late 1990s changes were made in the methods used to measure primary production, making it hard to know if apparent increases after 1998 are real (Rydberg, Ærtebjerg, and Edler 2006). Conley et al. (2007) carried out a detailed statistical analysis of bottom water oxygen concentrations in Danish estuaries and open waters, to see if hypoxia was declining with decreasing nutrient loading and productivity. The result is instructive. While declining N loading appeared to be correlated with increasing oxygen concentrations in bottom waters during summer (as expected), declining wind speed and increasing water temperature combined to produce net declines in bottom water oxygen, and no improvement was realized. Although it follows that conditions would have been worse in the absence of the nutrient reduction, it is disappointing not to have found a more positive response to the management effort.

If eutrophication is an increase in the supply of organic matter to an ecosystem, a decline in the supply of organic matter is called "oligotrophication" (Nixon, in press). This phenomenon has received increasing attention in lakes, where nutrient pollution and eutrophication attracted management interest at least a decade earlier than in coastal marine ecosystems (e.g., Nay 1996; Anderson, Jeppesen, and Soendergaard 2005). Oligotrophication has received almost no attention in marine ecology, but this will surely change as management actions take effect. In some cases, the results of oligotrophication may be disappointing, as with hypoxia in the Baltic Sea or the Seto Inland Sea off Japan, where fish landings appear to have declined with nutrient reductions (Yamamoto 2003). In other cases, it may prove difficult to document cause and effect relationships. A case in point is the Dutch Wadden Sea, where extensive monitoring over many decades has shown a complex and somewhat confusing response to reduced nutrient loading (Philippart et al. 2007). While phytoplankton biomass and the productivity of both phytoplankton and phytobenthos increased markedly with increasing nutrient enrichment during the 1970s and early 1980s, declines in phytoplankton production were more modest following nutrient reduction, and total biomass remained relatively constant. However, the contribution of diatoms to biomass declined markedly with nutrient reduction. The complex interplay of "bottom-up" (nutrient enrichment) and "top-down" (grazing) processes made it difficult to correlate ecosystem changes, especially of upper trophic levels, with nutrient reduction. After assessing benthic animals and marine birds, Philippart et al. (2007) concluded:

> In contrast to the sequential increase in biomass of phytoplankton and macrozoobenthos during nutrient enrichment … subsequent nutrient reduction affected the biomass of these communities to a much lesser extent. The weak coupling between nutrient levels and biomass during the reduction phase might be a result of a delayed response … and concurrent changes in species composition … which can dampen the numerical and biomass responses at higher trophic levels.

It is a characteristic of complex systems that their history is an important influence on their future behavior, and we should not expect the path of oligotrophication to trace in reverse the exact steps of eutrophication. A further complication to predicting the response of coastal marine ecosystems to nutrient reduction and/or oligotrophication is that many other factors influencing the behavior of the ecosystem will almost certainly have been changing during the time of nutrient enrichment. Carlos Duarte and colleagues (2009) have assembled data on chlorophyll (as a measure of the biomass of phytoplankton) from a number of coastal systems that experienced nutrient enrichment followed by nutrient reduction. In no case did chlorophyll concentrations simply retreat with declining nutrient inputs along the same trajectory they followed while increasing during nutrient enrichment. They caution that, because of shifting baselines, managers (and scientists) who expect to restore coastal systems to a prior state simply by reducing nutrient inputs are trying to "Return to Neverland", home of the mythical Peter Pan and the Lost Boys. For these reasons, we must expect many surprises in the future from the temperate estuaries that have received so much of our attention (and our nutrients) in recent decades. And we cannot lose sight of the larger picture, that oligotrophication, like eutrophication, may be caused by factors other than changes in nutrient inputs. For example, two decades of oligotrophication in Narragansett Bay, Rhode Island (United States) appear to have been the result of increased temperature and clouds during a time of relatively stable nutrient inputs (Li and Smayda 1998; Fulweiler et al. 2007; Nixon et al. 2009; Fulweiler and Nixon, in press). And the supplies of nutrients themselves are influenced strongly by large-scale changes in climate and hydrography that may alter the carrying capacity of the environment; one of the most striking examples of which may be the decline of marine mammals and benthic animals with the climate-induced oligotrophication of the Bering Sea (Schell 2000; Grebmeier et al. 2006 respectively).

We close with a final observation that nutrient pollution lies at the intersection of two of the major themes of coastal ecology: the causes of productivity, and the impacts of pollution. It is not surprising that the topic embraces complications and conflicts. The eutrophication that nutrient pollution often causes is a fundamental change in the economy of the ecosystem, and it is not clear that the lessons we have learned from four decades of study in temperate coastal systems will hold as the very low nutrient waters of the tropics become enriched. For example, recent work off the Nile Delta has shown that anthropogenic nutrients may stimulate fisheries productivity (Oczkowski et al. 2009). Studying and managing nutrient pollution and eutrophication in tropical coastal environments is a major and immediate challenge for marine ecology. While it seems virtually certain that the world faces a future in which the cycles of N and P become increasingly perturbed, there are some reasons for optimism. The evidence from Europe, North America, and Japan is that as societies get richer, they invest more in pollution abatement. Hence, as the growing wealth of developing nations allows them to eat more meat and use more fertilizer, it may also allow them to invest more in the education and infrastructure that can mitigate nutrient and other forms of pollution. Moreover, because of the links between nutrient pollution and other environmental threats that we discussed earlier, many actions that may be taken to reduce carbon dioxide emissions and acid rain will also help to reduce N pollution. Actions taken to protect wetlands and riparian zones, due to their habitat values for wildlife, will also make watersheds hold or remove reactive N and P. To the extent that campaigns to improve human diet through education are successful, they will also reduce nutrient pollution. As the great limnologist G. E. Hutchinson (1969) pointed out, the term "eutrophic" was used in medicine to mean "well-nourished", long before it was taken up by ecologists. If the human population really becomes "eutrophic" by eating less meat and animal fat, it will go a long way to protecting the coastal marine environment from eutrophication.

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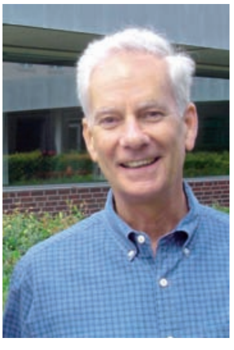

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