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Potential for Deliberate Management of Element Interactions to Address Major Environmental Issues

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Human activities often decouple elements from their natural stoichiometry through the selective release and mobilization of elements such as C, N, S, and P from their long-term stores (Austin et al., Chapter 2, this volume). Many environmental problems arise from these biogeochemical changes, including rising atmospheric CO_2 and the degradation of the quality of land, air, and water. A mechanistic understanding of elemental interactions provides an opportunity for deliberate manipulations of elements in order to mitigate environmental problems. In this chapter we demonstrate that recoupling of elements can be a successful strategy for minimizing or alleviating the harmful effects of human activities and for restoring degraded ecosystems.

There are two fundamentally different pathways by which elemental interactions can be manipulated to achieve environmental goals. The first is to limit the source of the excess elemental inputs (limiting primary forcing). Examples include decreasing the emissions of pollutants or greenhouse gases or minimizing the applications of fertilizers or other chemicals. A second approach is to minimize the effects of excess elements by introducing another change to counteract the undesirable effects of the primary forcing. Examples of enhanced secondary forcing would be liming of acidified lakes and soils in order to mitigate acidification or enhancing carbon sequestration by fertilizing terrestrial systems with nitrogen and oceans with iron.

Minimizing the direct source of an environmental problem is the most desirable and effective approach. Even though this approach is often relatively straightforward, an understanding of element interactions may be critical for successful mitigation of a problem. For example, reductions in tropospheric ozone (O_3) concentrations by minimizing emissions of volatile organic carbon (VOC) were less efficient than expected, particularly in urban environments, because ozone production was frequently limited by concentrations of nitrogen oxides (NO_x) rather than by VOC (Chameides et al. 1988; NRC 1991).

Mitigation of environmental problems by enhancing secondary forcing is a much less reliable option. Manipulation of the environment, even with the aim to restore it, involves inevitable risks of side-effects that could be difficult to foresee. Manipulating elemental interactions might cure one environmental problem but also create or intensify another. Thus, a broad view needs to be taken in order to evaluate the overall effects of a manipulation on multiple aspects of the environment. To illustrate the differences between the two approaches, we will examine the case of lake acidification. Reductions in the deposition of acidic compounds will likely allow for the recovery of lakes and downstream ecosystems (Galloway, Chapter 14, this volume). Decreases in acid deposition will also improve air quality and acidified soils in the lake's catchment. On the other hand, liming of lakes might be equally effective in restoring lake pH. The extinct organisms may not be brought back by liming, however, and the relative abundance of species established in the lake after the liming might be different from pre-acidification because of the different pathway by which the lake chemistry was achieved. Furthermore, by not treating the source of the problem, the lakes will need to be limed indefinitely and other efforts will be needed to address the negative effects of acid deposition on, for example, air quality, soils, corrosion, and human health. From the environmental point of view, it is clearly more efficient and effective to limit the primary forcing by reducing emissions of pollutants that cause the acidification rather than employing ecosystem manipulations that will address only a subset of the problem and may contribute to other environmental problems.

The popularity of the enhanced secondary forcing approach is more political than ecological. And although it has been argued that limiting the source of the problem would have negative economic impacts, the suite of ecosystem manipulations that are necessary to secondarily address the problems of increased pollution emissions are likely much more expensive and less effective. Minimizing the primary forcing, although desirable, has its limits set by the expanding needs of the world's population (Figure 5.1). Enhanced secondary forcing might be necessary if parts of the environment are to be protected. It could also help to preserve the environment over critical periods of time between when a problem is recog-

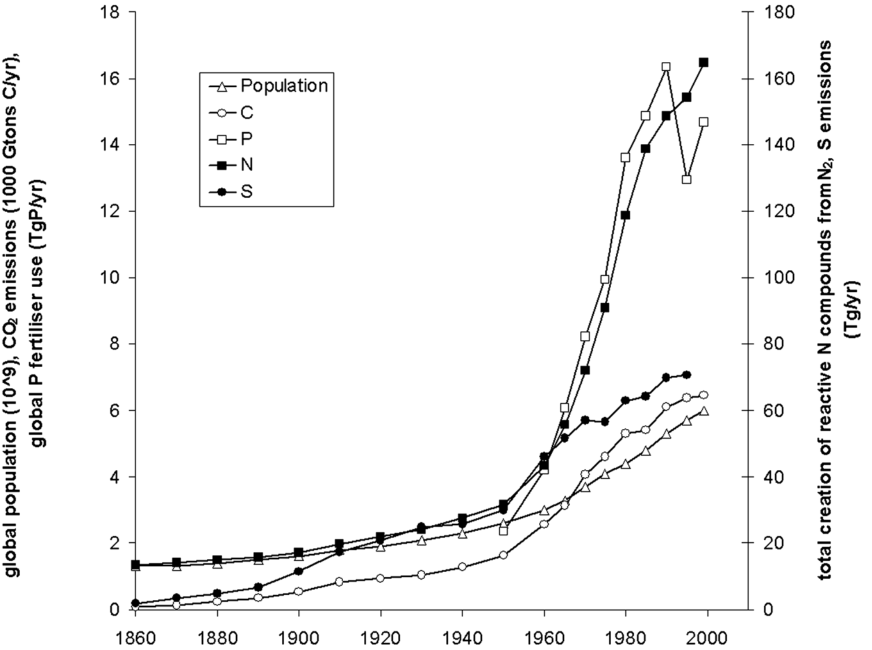

Figure 5.1. Global population (billions), total CO_2 emissions from fossil fuels (1,000 Gtons C/yr), global P fertilizer consumption (Tg P/yr), and total creation of reactive N compounds from N_2 by fossil-fuel combustion, cultivation-induced biological N_2 fixation, and the Haber-Bosch process (Tg/yr). Data on CO_2 emissions from fossil fuel combustion, P fertilizer consumption, and SO_2 emissions from fossil fuel combustion and sulfide ore-smelting were provided by May Ver (University of Hawaii) and reported upon in Mackenzie, Ver, and Lerman (2002). Specific sources of the data were: CO_2, Marland et al. (2002); P, FAO (1950–1999); S and N, Dignon and Hameed (1989), Dignon (1992), Brown, Renner, and Flavin (1997), and Galloway et al. (2003).

nized and when the cause of the problem is under control. Understanding element interactions enables us both to limit the primary forcing and to minimize damage through secondary forcing mechanisms. Which of the two pathways (or what combination of the two) is the best strategy will vary for different environmental problems.

In this chapter we address the potential for deliberate element manipulations both to limit the primary forcing and to enhance the secondary forcing in a broad sense, with some examples on scales from local to global and from different environments and different parts of the globe. The addressed environmental problems are divided into four areas:

1. air, water, and soil quality degradation due to emissions of nitrogen and sulfur
2. increased greenhouse gas emissions
3. nutrient enrichment of coastal ecosystems
4. land degradation

Air, Water, and Soil Quality Degradation Due to Emissions of Nitrogen and Sulfur

Energy production results in the emission of NO_x and SO_2 to the atmosphere. Food production results in the emission of S (H_2S, organic S) and N (NH_3, N_2O, NO, organic N) species to the atmosphere. Large regions of all continents (except Antarctica) have experienced significant increases in these emissions over the past few decades, and there are projections for significant increases in the future, especially in the developing world.

Agroecosystems are defined as cropland and animal production systems (i.e., animal feeding operations, or AFOs, and pastures). Both fertilized fields and AFOs are large sources of NH_3, N_2O, and NO emissions to the atmosphere, as well as N and P losses to aquatic ecosystems. The N and P budgets for an agroecosystem can be described by its inputs, its internal recycling, and its outputs. These outputs are products and losses to water and, for N, to air (Figure 5.2). These losses have negative impacts on both the farm economy and the environment and should be reduced as much as possible in a financially sustainable way.

These emissions to air and water have consequences for the health of people and aquatic and terrestrial ecosystems. SO_2 is converted to sulfate aerosol in the atmosphere, where it reduces radiation and negatively impacts human health. Once deposited to the Earth's surface, it has the potential to acidify both soils and freshwater ecosystems (Galloway, Chapter 14, this volume). N_2O is a greenhouse gas in the troposphere and contributes to stratospheric ozone depletion. The other N species also have environmental impacts. NO_x contributes to increased levels of O_3 in the troposphere (Finlayson-Pitts and Pitts 2000), which can result in decreased productivity of terrestrial ecosystems (Ollinger et al., Chapter 4, this volume) and have significant human health effects (Follet and Follet 2001; Wolfe and Patz 2002). NH_3 emitted to the atmosphere can react with sulfuric acid creating an ammonium sulfate aerosol. NH_3 plays an important role in the direct and indirect effects of aerosols on radiative forcing and thus global climate change (Penner, Robinson, and Woods 2001; Seinfeld and Pandis 1998).

In addition to these direct effects, NO_x emissions contribute to a wide variety of other environmental impacts as they are converted to other chemical species and cycle through environmental reservoirs. Referred to as the "nitrogen cascade" (Galloway et al. in press; Galloway, Chapter 14, this volume) and the "phospho-

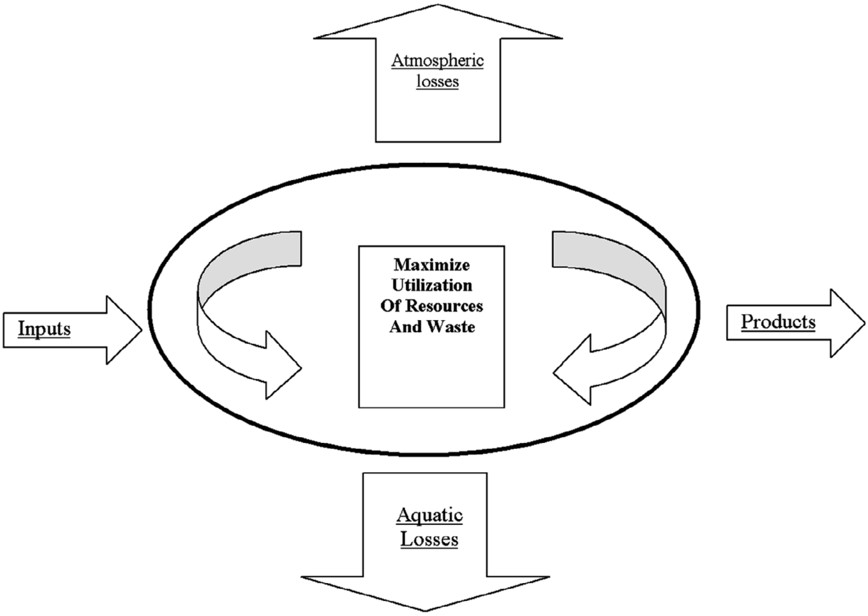

Figure 5.2. Schematic diagram of animal feeding operations (AFO), animals plus associated cropland. Adapted from NRC (2003).

rus cascade" (Newbold et al. 1983), the sequential transfer of reactive N and P through environmental systems results in environmental changes. N and P move through or are temporarily stored within each system, where they contribute to terrestrial N saturation (Aber et al. 1998), biodiversity losses (Eviner and Chapin, Chapter 8, this volume), and freshwater and coastal eutrophication (Rabalais 2002).

Element Interactions That Can Be Used to Address the Issue

EMISSIONS OF NO_x AND SO_2 FROM FOSSIL FUEL COMBUSTION

Understanding the interactions of SO_2 and NO_x with other chemical species has provided the basis for the development of the technology to reduce SO_2 and NO_x emissions to the atmosphere by treatments before, during, or after fossil fuel combustion. SO_2 is removed by reaction with $CaCO_3$ producing $CaSO_4$. NO_x is converted to N_2 by catalytic reaction with NH_3. These technologies have been extensively used in Europe and to a lesser degree in North America to cause substantial reductions in SO_2 and NO_x emissions. Although the application of the technology in developing regions is less common, in some regions SO_2 emissions have been extensive owing to change in fuel type (e.g., China). A major barrier to applying the existing technology in developing regions is cost. To address this issue, current research using our knowledge of elemental interactions is investigating how to make reductions in NO_x emissions more effective and cheaper (e.g., Bradley and Jones 2002).

Emissions from Agroecosystems

Element interactions can be manipulated through a number of pathways in agroecosystems to reduce the losses of reactive N and P to the environment, to increase the recycling of N and P within the agroecosystem, and thus to increase the efficiency of N and P use in agroecosystems. Careful selection of crops and proper composition and timing of fertilizer application and irrigation can maximize crop uptake of available N and P (Matson, Naylor, and Ortiz-Monasterio 1998). More frequent, smaller inputs of fertilizer that are timed with plant demand are more likely to be taken up by plants and microbes and less likely to be lost to runoff or to the atmosphere. N retention of fertilizer additions can be greater from organic fertilizers, which have gradual N release, with the C:N ratio of the organic fertilizer influencing the rate of release. As with inorganic fertilizers, careful attention must be given to the amount and timing of manure application to minimize nutrient losses. Similarly, slow-release inorganic fertilizers, such as S-coated urea, can minimize leaching losses. Nitrification inhibitors can be used with N fertilizers to minimize N losses by NO_3^- leaching. Plant compounds such as phenolics have also been shown to inhibit nitrification, and nitrification can be minimized by growing these species as cover crops or adding their residues to the soil. Minimizing the spatial and seasonal prevalence of bare soil on croplands by cover cropping reduces leaching and erosion losses. In order to maximize the effectiveness of cover crops, plant species should be chosen based on desirable traits such as fast growth and extensive N and P uptake at the time of maximum leaching loss or at a time or depth different from the crop. The C:N ratio of cover crops can determine how quickly the cover crop residue is mineralized or to what extent these nutrients are immobilized by the microbial community.

In animal production systems, diet management ensures that the proper mix of N and P is present for uptake by animals, thus minimizing losses through excretion. Once animal waste is excreted, biogeochemical interactions may be used to maximize the loss of reactive N as N_2 to the atmosphere. For example, N emissions from AFOs could be reduced by using current knowledge of elemental ratios to create N_2 production systems that first maximize nitrification (aerobic systems) and then maximize denitrification to produce N_2 (anaerobic systems).

This approach would be especially appropriate where economic constraints limit waste recycling directly to agroecosystems (NRC 2003).

In a perfect management system all N and P added would be stoichiometrically matched to carbon and other elements in the production of crops or animals. The essence of the practices described is to use the minimum N and P necessary to support a given unit of productivity and to potentially decrease the losses of N and P to the water and atmosphere. The current management of conventional agroecosystems, however, makes some amount of loss inevitable.

Increased Greenhouse Gas Emissions

A serious environmental problem that has received significant attention is the accumulation of carbon dioxide and other radiatively active gases in the atmosphere, which in turn increases the surface temperatures of the world. Limitation of greenhouse gas emissions is the most direct means to reduce atmospheric increases. But because anthropogenic emissions of CO_2, CH_4, and N_2O are directly tied to critical economic activities of food and energy production, strong source controls are politically contentious and difficult to implement. Future increases predicted for the global population and increasing "industrialization" confound attempts to reduce emissions. Even though strong source controls are controversial, it should be noted that there is substantial potential for limiting emissions by conserving energy and by using environment-friendly technologies without significantly affecting lifestyle. An alternative or a complement to limiting the primary forcing by emission controls is the enhancement of sinks, particularly for CO_2, by forcing natural ecosystem processes to enhance the long-term retention of carbon. Understanding of the interaction of elemental cycles and the interactions of mainly C, N, and P will be critical to the management of greenhouse gases. Developing viable long-term solutions is a formidable challenge, as will be discussed.

Element Interactions That Might Be Used to Address the Issue

TERRESTRIAL ECOSYSTEMS

In terrestrial ecosystems, carbon storage will depend upon the balance of net primary productivity (NPP) and heterotrophic respiration (Rh) (Randerson et al. 2002). Thus, CO_2 sequestration may be increased either by augmenting NPP or reducing the rate of Rh. A number of element interactions potentially come into play in controlling this balance. For a given area NPP may be limited by availability of energy (radiation), water, and/or of one or several nutrient elements. In

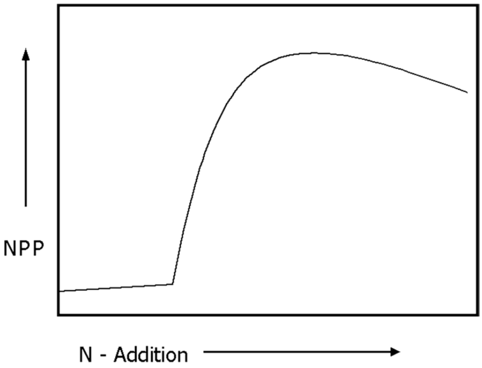

Figure 5.3. Conceptual relationship between net primary production (NPP) and N addition. Adapted from Aber et al. (1989, 1998).

general, we cannot control radiation. Irrigation is used extensively to increase NPP for crops. Fertilization with nutrient elements (N, P, K, etc.), either intentionally or inadvertently (in the case of deposition of atmospheric pollutants), can also increase the NPP of managed and unmanaged terrestrial ecosystems.

Nitrogen is the main limiting nutrient for primary production in many temperate terrestrial ecosystems (Ågren et al., Chapter 7, this volume). In Figure 5.3 we illustrate a theoretical response of productivity to a given level of nitrogen addition. Systems on the left-hand side of the graph are severely nitrogen limited, and these respond to nitrogen addition first by immobilization of added N. The degree to which the added N is partitioned between plants, decomposers, and nonliving organic matter will depend upon the form and timing of the addition, as well as plant species composition (Holland and Carroll, Chapter 15, this volume). NPP will increase with N addition until it reaches a level where it is no longer limiting. Beyond that level, the system will become nitrogen saturated (Aber et al. 1989, 1998) and productivity will decline.

Enhanced sequestration of C in forests associated with increased N inputs has received considerable attention (Schindler and Bayley 1993; Nadelhoffer et al. 1999). The relationship between N inputs and C sequestration, however, is complex. We suggest a framework based on interactions between N and C that may explain responses of decomposition to N additions. A number of studies have found that high availability of soil inorganic N stimulates decomposition of recalcitrant litter and old soil C. In contrast, additions of high-N litter tend to lower utilization of recalcitrant C compounds (Billes, Gandaisriollet, and Bottner 1986; van Ginkel, Gorissen, and Van Veen 1996). While initial decomposition rates of high-N litter are rapid, total mass loss is often lower for high-quality litter than for low-quality litter, with high-N litter making a greater contribution to soil

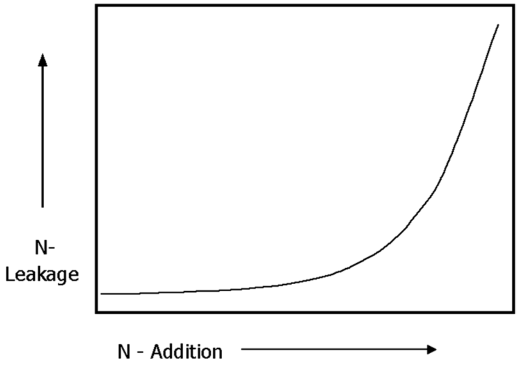

Figure 5.4. Conceptual relationship between N leakage and N addition. Adapted from Aber et al. (1989, 1998).

organic C storage (Berg 2000; Eviner 2001). The effects of N additions on decomposition will also depend on labile C availability to microbes. Labile C can stimulate decomposition of recalcitrant litter and soil organic C under low nutrient conditions, since microbes use labile C as an energy source to metabolize recalcitrant C in search of N. In the presence of high amounts of N, microbes utilize the available C and N and do not decompose more recalcitrant substrates (van Ginkel, Gorissen, and Van Veen 1996).

Based on these experimental results, we hypothesize that when microbes have access to easily available N and C, they will not utilize recalcitrant sources. This hypothesis should apply whether they are presented with these in the form of labile litter, or labile C and inorganic N in soil solution. The presence of either labile C or inorganic N, without the other, will stimulate microbial utilization of more recalcitrant substrates, both as litter and soil organic C. Available C and N in high-quality litter is quickly utilized, but the remaining C in this litter becomes relatively stable. This scenario may be complicated by shifts in the microbial community, which differentially mediate interactions between these elements. In addition, protozoal grazing of rhizosphere bacteria may release N and alter interactions between N and C. Thus, the effect of nitrogen additions on decomposition can vary depending on the amount of nitrogen added, its fate in plant material versus inorganic soil pools, and its influence on labile C through influencing exudation of a species or changing species composition.

There are also possible trade-offs from adding N to sequester C. Additions of N to increase NPP will also increase N leakage from the ecosystem (Aber et al. 1989). Depending upon environmental conditions, N losses may occur in forms such as NO_3^- (lost to groundwater) or N_2O (emitted to the atmosphere). Figure 5.4 illustrates the proportion of N leakage versus N applied. In highly N-limited systems, N leakage will be minimized and the dominant form of N loss will be dissolved organic nitrogen (DON) (Perakis and Hedin 2002). At some level of N addition, leakage of more environmentally sensitive forms such as NO_3^- and N_2O will occur. The leakage will increase, probably nonlinearly, through and beyond the level of N saturation.

AQUATIC ECOSYSTEMS

Because of their vast volume, world oceans are frequently considered a long-term sink for the increasing CO_2 in the atmosphere. One proposal has been to fertilize the ocean with a limiting nutrient (e.g., iron, or Fe) and thereby stimulate NPP. Assuming vertical flux of carbon to a depth below 3,000 meters, the C from the atmosphere could hypothetically be retained for periods of about 1,000 years. Various regions of the ocean (e.g., the Southern Ocean, the North Pacific, and the South Pacific) are iron-limited (Ducklow, Oliver, and Smith, Chapter 16, this volume), and trace additions result in increased production and biomass accumulation. Because of elemental interactions, however, there are problems associated with enhancing the secondary forcing by ocean fertilization. Removal of iron limitation will result in nitrogen, silica (Si), or phosphorus limitation, thus restricting enhancement of NPP in some regions. Furthermore, secondary limitations may induce significant planktonic assemblage changes, which will then influence export and food web transformations. Addition of significant amounts of Fe to stimulate productivity in the Southern Ocean may result in the generation of anoxic waters at 300–800 meters (Sarmiento and Orr 1991), which would disrupt the extant food web of the region. Most (99.9 percent) of the surface production is remineralized in the upper 3,000 meters (Suess 1981), and hence natural carbon sequestration is a relatively inefficient process. Finally, even if all available surface nutrients were utilized to produce biomass, ocean sequestration of carbon would only decrease *the rate of increase* in atmospheric CO_2 by 10 percent (Sarmiento et al. 1998). Given that surface nutrients cannot be completely removed due to elemental interactions (e.g., Si limitation can preclude complete removal of N and P), the quantitative effect of forced oceanic carbon sequestration is small relative to emission increases.

Freshwater systems store a proportionately large amount of carbon in sediments by virtue of the large bottom area-to-volume ratio and sediments that are highly enriched in particulate organic carbon (POC). Therefore, even though lakes and reservoirs have relatively small areas, they can contribute significantly to carbon sequestration (Dean and Gorham 1998). Carbon turnover in these systems is faster than in the ocean (because of the finite lifetime of most reservoirs and shallow lakes), and so this C storage would be more short term (hundreds of years).

Nutrient Enrichment of Coastal Ecosystems

Nutrient inputs to coastal ecosystems around the world have markedly increased, as discussed in this volume by Austin et al. (Chapter 2). The environmental consequences of nutrient enrichment of coastal ecosystems are now well documented (e.g., Smetacek et al. 1991; Vitousek et al. 1997; NRC 2000; Cloern 2001; Rabalais 2002) and include increased phytoplankton production, increased turbidity with subsequent loss of submerged aquatic vegetation, oxygen deficiency, decreased biodiversity, and changes in species composition and food web structure. The specific effects on a coastal ecosystem depend on a number of factors including both the quantity and quality (forms and ratios) of nutrient inputs (Seitzinger et al. 2002a). The widespread alteration in elemental inputs due to human activity is likely to continue in the future (Kroeze and Seitzinger 1998; Austin et al., Chapter 2, this volume).

Major sources of N and P to coastal ecosystems include agroecosystems, human sewage, and, for N, atmospheric deposition. A portion of the N and P from these sources in watersheds is sequentially transferred to groundwater, riparian zones, wetlands, rivers, and coastal marine ecosystems. At each step along the way N and P can be temporarily stored or permanently removed within each system. This sequential transfer has been referred to as a P cascade (Newbold et al. 1983) and more recently applied to N as an "N cascade" (Galloway et al. 2003; Galloway, Chapter 14, this volume).

Element Interactions That Might Be Used to Address the Issue

Once nutrients are lost from terrestrial systems, there is the potential for nutrient removal at a number of locations along the nutrient cascade before they reach coastal systems. Knowledge of element interactions can be used to optimize nutrient removal along this path. For example, the selection of plant species to store nutrients and influence nutrient recycling rates can be important in transition zones between agricultural lands and aquatic systems, such as riparian strips. Although riparian strips can reduce total P export by 98 percent (Martinelli, Chapter 10, this volume), bioavailable P load may increase in some cases because of leaching from standing vegetation. This possibility highlights the importance of mechanisms other than plant uptake in providing effective control of nutrients. Harvesting aboveground vegetation in buffer zones can remove nitrogen and phosphorus and prevent possible nutrient loss to aquatic systems through leaching or mineralization from residues. This step can be especially important in buffer zones that may become N saturated over time. This solution may not be sustainable, however, since biomass harvest will remove a significant amount of other nutrients over time, impacting the long-term ability of these transition zones to take up nitrogen because of limitation of plant growth by other elements.

Wetlands are another transitional ecosystem that can influence export of nutrients downstream to coastal ecosystems. Wetlands can be important zones of denitrification, and thus remove soluble inorganic N from the water. Denitrification, however, may result in release of N_2O, a potent greenhouse gas. Plant species in wetlands can have significant effects on the conditions necessary for denitrification, including oxygen levels and the presence of organic C. High organic C availability per unit of NO_3^- is key to driving denitrification to N_2, so planting species with high labile C inputs could minimize the trade-off between water and air quality. This approach, however, can also favor dissimilatory nitrate reduction to ammonium, which would preserve N in aquatic systems.

Wetlands may decrease inorganic N inputs to aquatic systems, but they often increase organic N inputs. While these organic inputs are commonly only 20 percent or less bioavailable, they still contribute to coastal eutrophication (Stepanauskas, Leonardson, and Tranvik 1999; Wiegner 2002). Although wetlands may play an important role in minimizing N inputs to aquatic systems, there are trade-offs in management for N versus P. An oxidized soil environment is most likely to cause sorption and precipitation of P on Fe oxides. Under the reducing conditions that cause denitrification and reduce N transfers, Fe^{3+} will be reduced to Fe^{2+} and release bound phosphate to percolating water. Although denitrification occurs at a higher pH than the conversion of Fe^{3+} to Fe^{2+}, it is impossible to manage the reducing potential of riparian zone soils closely enough to control both N and P transfers or retention.

Once N and P enter streams and rivers, a number of factors can affect their transfer to coastal ecosystems. Hydrology is one. For example, at the watershed scale, approximately 30–70 percent of the N inputs to the stream/river network can be denitrified (Seitzinger et al. 2002c). Channeling rivers tends to decrease the residence time of water, leading to a decrease in the interactions of solution and particulate N with the sediments, thus decreasing denitrification rates. Transfer of N and P to terrestrial systems (such as floodplains) also is minimized by channelization, due to faster water flux and less contact with floodplains. Reservoirs in rivers increase water residence time within that reach and thus can increase removal of both N (denitrification, particulate burial) and P (particulate burial). At the watershed scale the effect of reservoirs on nutrient export to coastal systems is highly dependent upon where in the watershed the reservoirs are located and how much of the total N and P pass through the reservoirs (Seitzinger et al. 2002c). Reservoirs are also a sink for dissolved Si.

Nutrients that are not retained within terrestrial ecosystems or removed along the nutrient cascade in riparian zones, wetlands, rivers, and reservoirs enter coastal ecosystems. There they contribute to a wide range of environmental problems, as already discussed. Although we are beginning to understand factors that can reduce N, P, and/or Si within ecosystem types, considerable additional knowledge of element interactions is required to effectively optimize nutrient retention within watersheds to reduce effects of nutrients in coastal systems. For example, within coastal ecosystems advancement in knowledge is needed on which nutrients (N, P, Si) and what nutrient forms (inorganic, organic) are limiting biological productivity over space and time and what the quantitative relationships between nutrient inputs are (quantities, forms, and ratios between them) and the effects on coastal ecosystems. This information must be coupled with advancements in knowledge of the relative magnitude of N, P, and Si (by form) removal within ecosystem types within watersheds and factors controlling their removal. This information then needs to be integrated at the watershed scale. Approaches are required that are applicable across a wide range of geographical regions because coastal eutrophication is a problem worldwide.

Land Degradation

Land use represents another form of primary forcing on the environment. Element loss, induced by decoupling biological elemental cycles from their natural stoichiometry through processes such as harvest, biomass destruction, and cultivation may lead to land degradation. Land use can also cause soil erosion, which is another major factor causing land degradation. In natural ecosystems mineralized N is rapidly incorporated into plant or microbial biomass, while only a small portion is lost by leaching or gaseous loss. P is largely conserved within the organic phase. Continued harvest of plant material removes these nutrients, and continued cultivation of an ecosystem requires the mobilization of N and P for crop export. Without this uncoupling of elements from the natural cycle, no production is possible unless nutrients are externally supplied. Many conventional farms in industrialized nations meet these nutrient demands through additions of inorganic fertilizers. Traditional shifting cultivation achieves this decoupling of nutrient elements by slash and burn. This practice results in substantial nutrient losses because most biomass C and N are released in the fire. P and cations are largely preserved in the ashes, but the vegetation-free soil will subsequently lose some of these elements through leaching or erosion. So although the decoupling of elements is essential to agriculture, the inefficient way in which this is accomplished in many systems causes large losses of organic matter and nutrients.

The effect of grazers is one of partial biomass destruction. Long-term overgrazing in grassland ecosystems not only changes the species composition and productivity through selective foraging and direct destruction, but also decreases soil fertility and changes the hydrological cycles (Dormar and Willms 1998; Wu and Tiessen 2002). In degraded grassland ecosystems of the world, soils are often prone to desertification because of diminishing organic matter input, accelerated evaporation, and destruction of surface soil structure. This desertification is usually accompanied by nutrient loss through surface soil erosion and nutrient leaching.

Finally, land degradation by direct pollution of the soil has become a critical issue in urban areas, mine sites, and other areas contaminated by toxic wastes.

Element Interactions That Might Be Used to Address the Issue

Avoiding the land degradation due to differential nutrient loss requires that elements be re-linked by slowing losses or facilitating the production of biomass and the accretion of organic matter. The concern over soil degradation and the shortage of productive land has prompted a search for alternatives that maintain N and P in organic matter and restore the balance of nutrients. This goal is being accomplished in a number of ways. In subhumid and semiarid environments the introduction of managed short fallows with a high proportion of leguminous trees or the planting of leguminous cover crops restore C and N budgets in the ecosystem (Jama et al. 2000). In humid tropical environments the use of chipped woody vegetation as a soil mulch supplies C and ties up soil N before it can be leached. In many farm operations, the use of plant residues and inorganic fertilizers successfully supplies nutrients, organic matter, and soil cover, restoring element linkages and preventing losses (Palm, Myers, and Nandwa 1997; da Silveira, Tiessen, and Tonneau 2001). Use of leguminous trees in the tropics (MacDicken 1994) in agroforestry and traditional farming systems is an effective way to maximize soil C input and to increase N input through N fixers, while simultaneously alleviating Al-toxicity and overcoming P deficiency (Haynes and Mokolobate 2001). This is an example of how local farmers empirically apply the principle of elemental interactions (C-N-P and Al) to stabilize yield, minimize risk, and maintain soil fertility in their traditional agricultural practices. These cultivation methods have evolved over centuries and can easily be seen in Mesoamerica, the Andean region, the Amazon Basin, and Asia (Altieri 1995; Pretty 1995).

In grazed ecosystems the maintenance of vegetation cover and ecosystem production is critical for protecting the system from the decoupling of nutrient interactions. Therefore, in degraded grassland ecosystems, remediation depends on limiting grazing impact by protecting land and destocking, thereby keeping the cycles of C, N, P, and S in biologically dominated processes rather than in chemically or physically controlled processes. Protection of these systems does not necessarily mean that grazing by animals is completely excluded. To some degree, free-ranging mammalian grazers can increase the availability of diet-enhancing nutrients (e.g., N and Na), thereby increasing their own carrying capacity (McNaughton, Banyikwa, and McNaughton 1997).

Legislation often requires remediation of polluted soils. Remediation is often accomplished by plants, which not only accumulate heavy metals but also promote the degradation of organic pollutants through the microbial community in their rhizosphere. Appropriate selection of plant species is key in successful bioremediation because of plants' differential ability to accumulate toxins (Banuelos 1996; Predieri et al. 2001), exude compounds that form stable complexes with contaminants (Balabane et al. 1999; Dousset et al. 2001), and enhance degradation through the exudation of labile C substrates (Anderson, Kruger, and Coats 1994; Liste and Alexander 1999, 2000). Nutrient imbalance of soil can markedly affect plant growth and also decomposition of recalcitrant pollutants. Similar to the substrate interactions influencing decomposition of plant residues and soil organic matter, the presence of easily degradable organic carbon can affect the degradation of recalcitrant organic pollutants. N inputs also influence this degradation, with high inputs of N fertilizers required in petroleum-polluted soils.

Conclusions

An understanding of elemental interactions is critical to effectively manage a variety of environmental problems. Managing ecosystems requires an understanding of the impact of elemental interactions on the pool sizes, rates, and timing of element cycling. Scientific theory often focuses on generalizations and broad-scale patterns, but management occurs at a local scale. In order for management to be effective, it must call on scientific theories that incorporate the specific interactions at that site and can explain the exceptions to the rule, not just the general trends. This is where a solid understanding of interactions between various elements at different concentrations is invaluable. For example, N additions can enhance microbial utilization of soil organic matter, but also can enhance the stabilization of organic matter. Understanding the role of interactions between N, C, and the microbial community is critical to manage soil C at a local scale.

Effective management of many elements also requires some guidelines for the desired balance of elements, the acceptable ranges of variation around this ratio, and the thresholds beyond which biogeochemical processes significantly change. For example, fertilization of crops or restoration sites by organic additions requires a C:N ratio that provides a balance of nutrient release and retention. Similarly, increasing the ratio of labile C to NO_3^- can enhance N_2 production during denitrification or, alternatively, can lead to dissimilatory nitrate reduction to ammonia and a retention of N in wetland and aquatic systems. Understanding the threshold at which this happens could be critical in managing N losses.

In many cases primary forcing has an impact at geographically distant areas. Air pollution can be transported over long distances. Emissions of greenhouse gases or depletion of stratospheric ozone are examples of environmental problems with global implications in which the geographical location of the sources of primary forcing plays a less important role. On a smaller scale the functioning of coastal ecosystems will depend on how effectively we can manage N, P, and Si export from terrestrial ecosystems. An example of the economic context at different scales is shown by slash-and-burn subsistence agriculture with shortened fallow periods that lead to land degradation. By definition, subsistence agriculture contributes only to the farm families' welfare and has no wider economic benefit. The land degradation resulting from inadequate management affects local and regional land quality. The greenhouse gas emissions produced by the slash and burn and subsequent soil organic matter mineralization are ultimately of global concern. Remedial action must occur in this context, and in the absence of on-farm resources, the regional or even global society will have to subsidize the remedial investment in sustainable land quality. This need is now recognized by major agricultural research and development agencies (Sanchez et al. 1997).

The difficulty of managing N losses in aquatic systems highlights another important management consideration. While many studies focus on the ecosystem effects of a small number of drivers at one time, systems must be managed simultaneously for multiple functions in response to multiple environmental changes. A knowledge of elemental interactions is key to evaluating the trade-offs of any management practice. For example, in order to minimize N input to aquatic systems, we might encourage denitrification. This step can have consequences for air quality, however, if N_2O is released. Similarly, in order to promote C storage, plant species with high litter quality may be utilized, while species with low litter quality are preferable for N immobilization. Because the cycling of elements is controlled by different factors, there will be trade-offs in managing for multiple elements. Ecologists need to highlight the overall impact of any element manipulation, because there are multiple ways to manage any element and the preferred method may be based on the trade-offs for other elements.

Ecology is often seen as a science that is distinct from, for example, physics and chemistry owing to its lack of general laws. Because ecosystems are a result of complex interactions, understanding the biogeochemical consequences of interactions of a large number of elements and environmental conditions, as well as the exceptions to the general patterns, greatly strengthens both the science of ecology and its application.

Overall Points

- Many environmental problems are a result of temporal and spatial decoupling of elements from the range of their natural balance.
- There are two pathways in which elemental interactions could be manipulated to achieve environmental goals. These are (1) limiting primary forcing and (2) enhancing secondary forcing.
- There are three main mechanisms for deliberate management of element interactions: (1) altering the inputs, (2) countering imbalances by selective additions of other elements, and (3) altering the species composition (to manage demand for individual nutrients, to recouple nutrients based on biotic stoichiometry, to overcome toxicity).
- Because elements interact, management practices that influence one elemental cycle will often influence others in different ways. Thus, any given management practice can have trade-offs or multiple benefits, and it is necessary to focus on the overall biogeochemical effects of a management practice to evaluate preferable methods.
- Presenting managers with the mechanistic framework that encompasses the range of biogeochemical interactions allows them to simultaneously manage multiple elements under variable local conditions.
- Element interactions have been effectively used to address environmental problems in a number of cases. In other cases current knowledge suggests that elemental interactions could be used more efficiently.

Specific Examples

Emissions of S and N

- While some of the environmental consequences of S and N emissions are clear, economic considerations often guide policy. Future research should consider the effectiveness and costs of mitigation efforts and compare these with the economic costs and environmental benefits of decreasing emissions and the costs of unmitigated S and N emissions. Such cost-benefit analyses are critical to guide policy decisions.
- Natural ecosystems have tight nutrient cycles linked to organic carbon. Agricultural use disrupts these tight cycles to make elements available for export and harvest but also makes the systems leaky. Recoupling of the nutrients to carbon is needed to reduce environmental impact.
- Where economic constraints limit waste recycling directly to agroecosystems, manipulation of elemental ratios, particularly through the use of coupled aero-

bic and anaerobic disposal systems, provides opportunities to limit waste impacts in general and specifically on denitrification.

Greenhouse Gas Emissions

- Stimulation of productivity by fertilization of both marine and terrestrial ecosystems has been proposed to decrease the net emissions of CO_2. In marine ecosystems the predicted maximum decrease in the rate of increase of atmospheric CO_2 that could be accomplished by Fe fertilizer is only 10 percent. In terrestrial ecosystems the relative effects of nutrient addition on productivity versus decomposition need to be considered.
- Total radiative impact associated with any management practice must be assessed by looking at the weighted production of all greenhouse gases (e.g., CO_2 sinks against N_2O and CH_4 sources).

Coastal Ecosystems

- Coastal eutrophication depends on controls of N, P, and Si in watersheds and in airsheds. Export of nutrients from terrestrial to aquatic systems can be minimized by optimizing crop utilization of applied nutrients, effectively managing manure from animal production systems, utilizing advanced wastewater treatment, and adopting technologies to reduce NO_x emissions from fossil fuel combustion.
- Reductions in nutrient export to aquatic systems can also be achieved by recoupling inorganic nutrients by integrating them with organic matter or by using elemental interactions to stimulate alternative loss pathways such as denitrification. Potential tools for these mitigation efforts are cover cropping, buffer strips, and wetlands. The quantity and stoichiometry of nutrient retention or export is a function of land use and land cover. Careful choice of the configuration of land use units in the landscape is key to managing nutrient losses.

Land Degradation

- Degradation of organic pollutants by microbial communities and heavy metal uptake by plants can be limited by nutrient availability.

Literature Cited

Aber, J. D., W. H. McDowell, K. J. Nadelhoffer, A. Magill, G. Berntson, M. Kamakea, S. G. McNulty, W. Currie, L. Rustad, and I. Fernandez. 1998. Nitrogen saturation in temperate forest ecosystems: Hypotheses revisited. *BioScience* 48:921–934.

Aber, J. D., K. J. Nadelhoffer, P. Steudler, and J. M. Melillo. 1989. Nitrogen saturation in northern forest ecosystems. *BioScience* 39:378–386.

Altieri, M. A. 1995. *Agroecology: The science of sustainable agriculture.* Boulder, Colo.: Westview Press.

Anderson, T. A., E. L. Kruger, and J. R. Coats. 1994. Enhanced degradation of a mixture of three herbicides in the rhizosphere of a herbicide-tolerant plant. *Chemosphere* 28:1551–1557.

Balabane, M., D. Faivre, F. van Oort, and H. Dahmani-Muller. 1999. Mutual effects of soil organic matter dynamics and heavy metals fate in a metallophyte grassland. *Environmental Pollution* 105:45–54.

Banuelos, G. S. 1996. Managing high levels of boron and selenium with trace element accumulator crops. *Journal of Environmental Science and Health.* Part A: *Environmental Science and Engineering and Toxic and Hazardous Substance Control* 31:1179–1196.

Berg, B. 2000. Litter decomposition and organic matter turnover in northern forest soils. *Forest Ecology and Management* 133:13–22.

Billes, G., N. Gandais-Riollet, and P. Bottner. 1986. Effect of Gramineae culture on the decomposition of C-14 and N-15-labeled litter in soil under controlled conditions. *Acta Oecologica-Oecologia Plantarum* 7:273–286.

Bradley, M. J., and B. M. Jones. 2002. Reducing global NOx emissions: Promoting the development of advanced energy and transportation technologies. *Ambio* 31:141–149.

Brown, L. R., M. Renner, and C. Flavin. 1997. *Vital signs: The environmental trends that are shaping our future.* New York: Worldwatch Institute.

Chameides, W. L., R. W. Lindsay, J. Richardson, and C. S. Kiang. 1988. The role of biogenic hydrocarbons in urban photochemical smog: Atlanta as a case study. *Science* 241:1473–1475.

Cloern, J. E. 2001. Our evolving conceptual model of the coastal eutrophication problem. *Marine Ecology Progress Series* 210:223–253.

da Silveira, L. M., H. Tiessen, and J. P. Tonneau. 2001. Organic matter management in family agriculture of semiarid Paraiba, Brazil. *Nutrient Cycling in Agroecosystems* 61:215–223.

Dean, W. E., and E. Gorham. 1998. Magnitude and significance of carbon burial in lakes, reservoirs, and peatlands. *Geology* 26:535–538.

Dignon, J. 1992. NOx and SOx emissions from fossil fuels: A global distribution. *Atmospheric Environment* 26:1157–1163.

Dignon, J., and S. Hameed. 1989. Global emissions of nitrogen and sulfur oxides from 1860 to 1980. *Journal Air Pollution Control Association* 39:180–186.

Dormar, J. F., and W. D. Willms. 1998. Effect of forty-four years of grazing on fescue grassland soils. *Journal of Range Management* 51:122–126.

Dousset, S., J. L. Morel, A. Jacobson, and G. Bitton. 2001. Copper binding capacity of root exudates of cultivated plants and associated weeds. *Biology and Fertility of Soils* 34:230–234.

Eviner, V. T. 2001. Linking plant community composition and ecosystem dynamics: Interactions of plant traits determine the ecosystem effects of plant species and plant species mixtures. Ph.D. diss., University of California, Berkeley.

FAO (Food and Agriculture Organization of the United Nations). 1950–1999. *Annual fertilizer review.* Rome.

Finlayson-Pitts, B. J., and J. N. Pitts. 2000. *Chemistry of the upper and lower atmosphere: Theory, experiments, and applications.* London: Academic Press.

Follett, J. R., and R. F. Follett. 2001. Utilization and metabolism of nitrogen by humans. Pp. 65–92 in *Nitrogen in the environment: Sources, problems, and management,* edited by R. F. Follett and J. L. Hatfield. Amsterdam: Elsevier Science.

Galloway, J. N., J. D. Aber, J. W. Erisman, S. P. Seitzinger, R. W. Howarth, E. B. Cowling, and B. J. Cosby. 2003. The nitrogen cascade. *BioScience* 53:341–356.

Haynes, R. J., and M. S. Mokolobate. 2001. Amelioration of Al toxicity and P deficiency in acid soils by addition of organic residues: A critical review of the phenomenon and the mechanisms involved. *Nutrient Cycling in Aroecosystems* 59:47–63.

Jama, B., C. A. Palm, R. J. Buresh, A. Niang, C. Gachengo, G. Nziguheba, and B. Amadalo. 2000. Tithonia diversifolia as a green manure for soil fertility improvement in western Kenya: A review. *Agroforestry Systems* 49:201–221.

Kroeze, C., and S. P. Seitzinger. 1998. Nitrogen inputs to rivers, estuaries, and continental shelves and related nitrous oxide emissions in 1990 and 2050: A global model. *Nutrient Cycling in Agroecosystems* 52:195–212.

Liste, H. H., and M. Alexander. 1999. Rapid screening of plants promoting phenanthrene degradation. *Journal of Environmental Quality* 28:1376–1377.

———. 2000. Plant-promoted pyrene degradation in soil. *Chemosphere.* 40:7–10.

MacDicken, K. G. 1994. *Selection and management of nitrogen-fixing trees.* Morrilton, Ark.: Winrock International Institute for Agricultural Development.

Mackenzie, F. T., L. M. Ver, and A. Lerman. 2002. Century-scale nitrogen and phosphorus controls of the carbon cycle. *Chemical Geology* 190:13–32.

Marland, G., T. A. Boden, and R. J. Andres. 2002. Global, regional, and national CO_2 emissions. In *Trends: A compendium of data on global change.* Oak Ridge, Tenn.: Carbon Dioxide Information Analysis Center, Oak Ridge National Laboratory, U.S. Department of Energy.

Matson, P., R. Naylor, and P. Ortiz-Monasterio. 1998. Integration of environmental, agronomic, and economic aspects of fertilizer management. *Science* 280:112–115.

McNaughton, S. J., F. F. Banyikwa, and M. M. McNaughton. 1997. Promotion of the cycling of diet-enhancing nutrients by African grazers. *Science* 278:1798–1800.

Nadelhoffer, K. J., B. A. Emmet, P. Gundersen, O. J. Kjønaas, C. J. Koopmans, P. Schleppi, A. Tietema, and R. F. Wright. 1999. Nitrogen deposition makes a minor contribution to carbon sequestration in temperate forests. *Nature* 398:145–148.

NRC (National Research Council). 1991. Rethinking the ozone problem in urban and regional air pollution. Washington, D.C.: National Academy Press.

———. 2000. Clean coastal waters: Understanding and reducing the effects of nutrient pollution. Washington, D.C.: National Academy Press.

———. 2003. Air emissions from animal feeding operations: Current knowledge, future needs. Final Report, Ad Hoc Committee on Air Emissions from Animal Feeding Operations, Committee on Animal Nutrition, Washington, D.C.

Newbold, J. D., J. W. Elwood, R. V. O'Neill, and A. L. Sheldon. 1983. Phosphorus dynamics in a woodland stream ecosystem: A study of nutrient spiralling. *Ecology* 64:1249–1265.

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. Pp. 193–

217 in *Replenishing soil fertility in Africa*, edited by R. J. Buresh, P. Sanchez, and F. Calhoun. SSSA Special Publication No. 51. Madison, Wisc.: Science Society of America.

Penner, M., C. Robinson, and M. Woods. 2001. The response of good and poor aspen clones to thinning. *Forestry Chronicle* 77:874–884.

Perakis, S. S., and L. O. Hedin. 2002. Nitrogen loss from unpolluted South American forests mainly via dissolved organic compounds. *Nature* 415:416–419.

Predieri, S., J. Figaj, L. Rachwal, E. Gatti, and F. Rapparini. 2001. Selection of woody species with enhanced uptake capacity: the case-study of Niedwiady resort pollution by pesticides stored in bunkers. *Minerva Biotecnologica* 13:111–116.

Pretty, J. 1995. *Regenerating agriculture*. Washington, D.C.: World Resources Institute.

Rabalais, N. 2002. Nitrogen in aquatic ecosystems. *Ambio* 31:102–112.

Randerson, J. T., F. S. Chapin, J. Harden, J. C. Neff, and M. Harmon. 2002. Net ecosystem production: A comprehensive measure of net carbon accumulation by ecosystems. *Ecological Applications* 12:937–947.

Sanchez, P. A., K. D. Shepherd, M. J. Soule, F. M. Place, R. J. Buresh, A. N. Izac, A. U. Mokwunye, F. R. Kwesiga, C. G. Ndiritu, and P. L. Woomer. 1997. Soil fertility replenishment in Africa: An investment in natural resource capital. Pp. 1–46 in *Replenishing soil fertility in Africa*, edited by R. J. Buresh, P. A. Sanchez, and F. Calhoun. SSSA Special Publication No. 51. Madison, Wisc.: Science Society of America.

Sarmiento, J. L., T. M. C. Hughes, R. J. Stouffer, and S. Manabe. 1998. Simulated response of the ocean carbon cycle to anthropogenic climate warming. *Nature* 393:245–249.

Sarmiento, J. L., and J. C. Orr. 1991. Three-dimensional simulations of the impact of Southern Ocean nutrient depletion on atmospheric CO_2 and ocean chemistry. *Limnology and Oceanography* 36:1928–1950.

Schindler, D. W., and S. E. Bayley. 1993. The biosphere as an increasing sink for atmospheric carbon: Estimates from increased nitrogen deposition. *Global Biogeochemical Cycles* 7:717–733.

Seinfeld, J. H., and S. N. Pandis. 1998. *Atmospheric chemistry and physics: From air pollution to global change*. New York: John Wiley and Sons.

Seitzinger, S. P., C. Kroeze, A. F. Bouwman, N. Caraco, F. Dentener, and R. V. Styles. 2002a. Global patterns of dissolved inorganic and particulate nitrogen inputs to coastal systems: Recent conditions and future projections. *Estuaries* 25:640–655.

Seitzinger, S. P., R. W. Sanders, and R. V. Styles. 2002b. Bioavailability of DON from natural and anthropogenic sources to estuarine plankton. *Limnology and Oceanography* 47:353–366.

Seitzinger, S. P., R. V. Styles, E. W. Boyer, R. B. Alexander, G. Billen, R. Howarth, B. Mayer, and N. van Breemen. 2002c. Nitrogen retention in rivers: Model development and application to watersheds in the northeastern U.S.A. *Biogeochemistry* 57:199–237.

Smetacek, V., U. Bathmann, E.-M. Nöthig, and R. Scharek. 1991. Coastal eutrophication: Causes and consequences. Pp. 251–279 in *Ocean margin processes in global change*, edited by R.F.C. Mantoura. New York: John Wiley and Sons.

Stepanauskas, R., L. Leonardson, and L. Tranvik. 1999. Bioavailability of wetland-derived DON to freshwater and marine bacterioplankton. *Limnology and Oceanography* 44:1477–1485.

Suess, E. 1981. Phosphate regeneration from sediments of the Peru continental margin by dissolution of fish debris. *Geochimica et Cosmochimica Acta* 45:577–588.

van Ginkel, J. H., A. Gorissen, and J. A. Van Veen. 1996. Long-term decomposition of grass roots as affected by elevated atmospheric carbon dioxide. *Journal of Environmental Quality* 25:1122–1128.

Vitousek, P. M., J. D. Aber, R. W. Howarth, G. E. Likens, P. A. Matson, D. W. Schindler, W. H. Schlesinger, and D. Tilman. 1997. Human alteration of the global nitrogen cycle: Sources and consequences. *Ecological Applications* 7:737–750.

Wiegner, T. N. 2002. Microbial and photochemical degradation of allochthonous dissolved organic matter within river systems and watersheds. Ph.D. diss., Rutgers, The State University of New Jersey.

Wolfe, A., and J. A. Patz. 2002. Nitrogen and human health: Direct and indirect impacts. *Ambio* 31:120–125.

Wu, R., and H. Tiessen. 2002. Effect of land use on soil degradation in alpine grassland soil, China. *Soil Science Society of America Journal* 66:1648–1655.