ECOLOGY

Controlling Eutrophication: Nitrogen and Phosphorus

Improvements in the water quality of many freshwater and most coastal marine ecosystems requires reductions in both nitrogen and phosphorus inputs.

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The need to reduce anthropogenic nutrient inputs to aquatic ecosystems in order to protect drinking-water supplies and to reduce eutrophication, including the proliferation of harmful algal blooms (*1*) and "dead zones" in coastal marine ecosystems (*2*) has been widely recognized. However, the costs of doing this are substantial; hence, developing the appropriate nutrient management strategy is very important. Nitrogen (N), needed for protein synthesis, and phosphorus (P), needed for DNA, RNA, and energy transfer, are both required to support aquatic plant growth and are the key limiting nutrients in most aquatic and terrestrial ecosystems. However, a cascading set of consequences has been set in motion, arising from massive increases in fixed N additions to the biosphere, largely through the production of fertilizers and increases in fossil fuel emissions (*3*). P levels have also significantly increased because of fertilizer use, as well as from municipal and industrial wastewater. Here, we explore the rationale for dual–nutrient reduction strategies for aquatic ecosystems, especially in estuarine and coastal marine regions.

The question of whether one or both nutrients should be controlled to reverse the detrimental effects of eutrophication of lakes was thought to be solved in the early 1970s by Schindler (*4*), who established that P was the primary limiting nutrient in remarkable long-term experimental manipulations at Lake 227 in the Experimental Lakes Area, Canada (*5*). These and other results (*6*) led to widespread reductions in P loading to North American and European lakes and consequent improvements in water quality (*7*). On the basis of lake examples, P controls were prescribed by environmental regulatory agencies for estuarine and coastal marine ecosystems as well (*8*). P-reduction programs improved water quality in many lakes, but broader water- and environmental-quality goals were not achieved, particularly in estuaries and coastal marine ecosystems. This led to the general recognition of the need to control N input to coastal waters (*9*).

Too much algae. (**Top**) Removing macroalgal blooms at the Olympic Sailing venue, China. (**Middle**) Seagrasses covered with attached algae in a Danish estuary. (**Bottom**) Non–N2-fixing cyanobacteria blooms in Lake Okeechobee, Florida, U.S.A.

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In lakes, the key symptom of eutrophication is cyanobacterial blooms (see figure, left). Planktonic N_2-fixing cyanobacteria bloom in fresh waters when P is replete and N availability is low. Such blooms are undesirable because cyanobacteria can be toxic, cause hypoxia, and disrupt food webs (*1*, *10*). N_2 fixation by cyanobacteria also can help to alleviate N shortages and hence maintain a lake in a P-limited condition (*5*).

N_2 fixation by planktonic cyanobacteria is much less likely in estuaries and coastal seas than in lakes. Significant coastal planktonic N_2 fixation has not been observed at salinities greater than 8 to 10 (ocean salinity is ~35), even in estuaries that are strongly N-limited, except in rare cases (*11*). If N limitation were the only factor governing blooms of N_2-fixing cyanobacteria, then their blooms would be widespread in estuarine and coastal marine ecosystems around the world, and they are not. Thus, reducing N loads to the saline waters of estuaries should not cause blooms of N_2-fixing cyanobacteria (*5*). Furthermore, estuarine and coastal marine ecosystem eutrophication results in loss of seagrasses and hypoxia (*2*), which are more serious recurrent problems than cyanobacterial blooms (see the figure).

Why is N_2 fixation in the saline waters of estuaries and coastal marine ecosystems so much less than that in lakes, and why is this process unresponsive to reduced N loads in estuaries? Numerous hypotheses have been put forward (*10*, *12*). Most researchers have concluded that no single factor is responsible, but rather interactions between two or more factors control the rates (*13*, *14*). Mesocosm experiments in Narragansett Bay (*12*) have indicated that the combination of slow growth rate from sulfate inhibition of molybdenum uptake and zooplankton grazing limited the accumulation of N_2-fixing cyanobacteria. Globally, significant N_2 fixation, particularly by the cyanobacterium *Trichodesmium*, does occur in the tropical and subtropical ocean,

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where denitrification depletes the available N and can be limited by P and Fe (*15*), but is not found in the more productive waters of estuaries and coastal seas.

P-only reduction strategies are likely to fail in Lakes Apopka, George, and Okeechobee, USA; Lakes Taihu and Donghu in China; and Lake Kasumigaura in Japan (*16*). In these lakes, P is rapidly recycled between sediments and water, and phytoplankton is dominated by non–N_2-fixing cyanobacteria, such as *Planktolyngbya*, *Oscillatoria*, and toxic *Microcystis*. *Microcystis* can vertically migrate, consume excess phosphorus at the sediment-water interface, and then rise to the water surface to form blooms (*10*). Careful simultaneous control of both P and N is required in such lakes to effectively control *Microcystis* and N_2 fixers.

Estuaries and coastal marine ecosystems that have been heavily loaded with nutrients can display P limitation, N limitation, and colimitation (*17*), and what nutrient is most limiting can change both seasonally and spatially (*18*). At the transition between fresh and saline water, P can often be the limiting nutrient (*17*, *19*). P and dissolved silicate are also often limiting during the spring, with N limitation commonly occurring during summer months (*18*). Algal production during summer is supported by rapidly recycled P within the water column or released from sediments. This condition is particularly true for coastal ecosystems, where the elevated salinity provides sulfate for microbial reduction in bottom sediments, which results in the release of large quantities of P (*19*). Also, although much of the P in freshwater systems is not biologically available because it is adsorbed by clay and other particles, a considerable fraction of the P desorbs as readily available, dissolved phosphate under saline conditions (*13*). Thus, as the summer progresses, available P increases as N declines and is not effectively compensated by N_2 fixation.

Pristine lakes are sufficiently different from estuarine and coastal marine ecosystems that they may be poor analogs (*5*). For example, the low-salinity conditions of the Baltic Sea present a complex situation where N_2 fixation does play an important role. The Baltic exhibits permanent bottom-water hypoxia (*20*), which increases N loss due to denitrification and anaerobic ammonium oxidation (annamox) at the interface between oxygenated and deoxygenated waters. The hypoxic conditions also result in injection of large amounts of P back into surface waters during deep winter mixing (*21*). The annual variation in sediment releases of P due to hypoxia is nearly an order of magnitude greater than the controllable P loads (*20*). Nitrogen has clearly been established as the nutrient limiting spring phytoplankton production; it is the sinking spring bloom that sends organic matter to bottom waters, which partly sustains hypoxia. The excess P in the water column leads to summer blooms of cyanobacteria, some of which are N_2 fixers that increase N concentrations in surface waters when they are abundant. This new N helps to sustain the springtime production and produces a "vicious circle" of eutrophication (*21*). Models suggest that, here, too, reductions in the inputs of both P and N are required for significant improvements in dissolved oxygen concentrations, transparency, and other water-quality conditions in the Baltic Sea (*22*).

Controlling only P inputs to freshwaters and ignoring the large anthropogenic inputs of N can reduce algal uptake of N and thus allow more N to be transported downstream where it can exacerbate eutrophication problems in estuarine and coastal marine ecosystems (*13*). For example, reductions in P loadings by improved wastewater treatment and banning the use of P-based detergents succeeded in arresting algal blooms in freshwater portions of the Neuse River estuary, North Carolina, USA, but increased eutrophication and hypoxia downstream in the estuary, where P is more rapidly recycled (*11*). Similarly, dramatic reductions in P loading from the Rhine River and other rivers draining into the North Sea before concomitant N reductions resulted in strong P limitation in the river plume, but greater N export, exacerbating eutrophication in waters of the Wadden Sea (*23*) and as far away as the Norwegian coast of the Skagerrak (*24*). Reductions in P loading have also been suggested to limit phytoplankton growth in the plume of the Mississippi River in the northern Gulf of Mexico, which was previously more strongly limited by N. However, the increasing and excessive loading of N, relative to P, from agriculture has driven the plume of the Mississippi River to periodic P limitation, especially during the spring bloom period (*25*). Implementing only P reductions without reducing N loads could displace the dead zone westward and increase its size (*26*).

It is prudent, and in most cases essential, to implement a dual–nutrient-reduction strategy when developing measures to control eutrophication. A focus on only P or N reduction should not be considered unless there is clear evidence or strong reasoning that a focus on only one nutrient is justified in that ecosystem and will not harm downstream ecosystems. Just as care should be taken to avoid reducing N inputs in a way that will increase compensating N_2 fixation, attention should also be given to avoid displacing the effects of eutrophication downstream by concentrating only on P in freshwater systems and avoiding watershed N reductions that can be very important for coastal marine ecosystems. Although some would suggest that management strategies that control one nutrient, such as the reduced use of fertilizers, handling of manure, soil conservation practices, and restoring wetlands and riparian buffers, would also control the other, this is not always the case (*27*). For example, the technologies for wastewater treatment to reduce P versus N differ markedly, and reducing atmospheric N deposition does not affect P inputs to aquatic ecosystems. Alleviation of eutrophication in aquatic ecosystems along the land-ocean continuum requires a balanced and strategic approach to control both nutrients appropriately.

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LETTERS

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Vines at risk. Symptoms of grapevine trunk disease.

The Spread of Grapevine Trunk Disease

THE GRAPEVINE DIEBACK DISEASES, ALSO CALLED GRAPEVINE TRUNK diseases, are the consequences of a complex of fungi that was described as early as the end of the 20th century. They attack the perennial organs of a vine and ultimately lead to the death of the plant. Over the past decade, the frequency of symptoms due to these fungi has considerably increased worldwide. For example, cumulated disease incidence values estimated for Italian vineyards may reach up to 50% (*1*). The lack of resources to fight the diseases and favorable environmental conditions worsen the situation. Sodium arsenic is the only treatment that has a potential effect against dieback diseases, but it has been prohibited in some countries. Some vineyards that have never been treated with sodium arsenic now present an exponential development of symptoms. Why are these symptoms emerging today? Are they due to changes in the vine behavior, in the climate, or in the microbial equilibrium, or are they due to undiagnosed pathogens? Despite all the studies on the fungi associated with the disease, on the host-pathogen interactions, and on the symptoms, the actual causes for their development are still elusive.

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Eutrophication: More Nitrogen Data Needed

WE AGREE WITH D. J. CONLEY *ET AL*. ("CONtrolling eutrophication: Nitrogen and phosphorus," Policy Forum, 20 February, p. 1014) that there are many compelling reasons for controlling agricultural and industrial sources of nitrogen. In many areas, nitrate and ammonium are now the main pollutants causing damage by acidification and base cation depletion in forests and freshwaters (*1*). In some areas, nitrate concentrations in drinking water have increased enough to exceed health standards (*2*). However, at this time, we cannot agree that reducing nitrogen is essential for controlling eutrophication, because there are insufficient whole ecosystem-scale data to show that removing nitrogen will reduce eutrophication.

Phosphorus control alone has succeeded in reducing eutrophication in many lakes [reviewed by (*3*)] and in at least one low-salinity estuary (*4*). In contrast, not a single ecosystem-scale study in any aquatic system has shown that reducing inputs of nitrogen decreases eutrophication.

The authors state that reducing phosphorus inputs has not reduced eutrophication in some lakes and many estuaries. In most cases, the reason is high "internal loading" of phosphorus from anoxic sediments. High concentrations of phosphorus and anoxia in surface sediments are the result of decades of high phosphorus loading causing increased settling and decomposition of organic matter. However, long-term studies of lakes in Europe (*5*, *6*) have shown that internal loading decreases slowly after external sources of phosphorus are controlled, so that ecosystems recover over a period of years to decades.

Many of the arguments put forward by Conley *et al*. are based on physiological or short-term indices of nitrogen limitation, which we have found to be spurious in our long-term, whole-lake manipulations. Simply put, over time algal and bacterial communities change to include species that fix nitrogen when fixed nitrogen limits the growth of other species. Small but long-term inputs of nitrogen through fixation and subsequent return from sediments eventually correct nitrogen deficits in ecosystems (*7*, *8*). The importance of these long-term, adaptive processes cannot be evaluated by short-term incubations or dissolved nutrient concentration ratios (*9*).

Conley *et al*. state that the nitrogen-fixing cyanobacteria that are capable of correcting ecosystem-scale nitrogen deficiencies in lakes are absent in saline estuaries. However, recent measurements of N_2/Ar ratios indicate that there is considerable N fixation by bacteria and phytobenthos in shallow, saline estuaries (*10*, *11*). Rates of fixation are similar to those that we have observed in lakes (*8*), and we anticipate similar results in overcoming N deficiencies. In fact, in whole systems, N removal accomplished at great expense in wastewater treatment may be offset by N fixation in natural open systems (*8*).

Globally, reducing inputs of nitrogen from sewage as well as phosphorus would require

Letters to the Editor

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spending many billions of dollars. The costs of removing both nutrients may even discourage any treatment in developing countries, particularly in the current economic depression. We believe that before the additional expense of nitrogen removal from sewage is to be imposed on society, it should first be demonstrated at ecosystem scales to effectively reduce eutrophication. **D. W. SCHINDLER[1]* AND R. E. HECKY[2]**

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Eutrophication: Focus on Phosphorus

THE POLICY FORUM BY D. J. CONLEY *ET AL.* ("Controlling eutrophication: Nitrogen and phosphorus," 20 February, p. 1014) advocates expensive and unnecessary nitrogen (N) control in lakes.

Many demonstrations of successful phosphorus (P)–only control in lakes are found in the literature (*1*). In the 1970s, P control was implemented in the Laurentian Great Lakes, an important North American freshwater source (*2*). Total P (TP) in Lake Ontario decreased to half the maximum in response to P management and by half again after zebra mussel invasion, reducing phytoplankton standing crop and shoreline nuisance blooms of Cladophora. The lower phytoplankton N demand alleviated nitrate shortages. Thus, Lake Ontario was a real-time experiment to validate P control as a means to manage eutrophication.

I take exception to Conley *et al.*'s prediction that P-only reduction strategies will fail in Lake Apopka. Sediments deposited since 1947 provide the basis for estimating whole-lake historic TP sedimentation (*3*). The sediment inventory shows average annual deposition of 0.367 g TP m^{-2} year^{-1} (1947 to 1996). Some TP, however, was in a form that is not readily recycled (*4*, *5*). External loading averaged 0.55 g TP m^{-2} year^{-1} in the 1990s when water-column TP was 0.320 g TP m^{-2} (*6*). The large TP sediment sink and short TP residence time in the water column indicate that sediment recycling is low (*6*). Therefore, strategies that control external P loading will control eutrophication over time. **CLAIRE L. SCHELSKE**

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Eutrophication: Model Before Acting

IN A RECENT POLICY FORUM ("CONTROLLING eutrophication: Nitrogen and phosphorus," 20 February, p. 1014), D. J. Conley *et al.* made a controversial case for a dual nutrient-reduction strategy to address eutrophication in lakes, estuaries, and coastal areas.

We believe that all asserted beneficial effects must be robustly predictable to ensure that society actually gets something in return for this effort, given the high cost of nutrient reductions. For instance, Swedish nitrogen (N) reductions in a very ambitious abatement plan for the Baltic Sea may not be possible to fulfill unless a large part of Swedish agriculture is permanently shut down, according to recent calculations by the Swedish Department of Agriculture (*1*).

Unfortunately, there are no general, validated mass-balance models for nitrogen that have been tested for independent coastal systems and been demonstrated to yield good predictive power. Any N model can be tuned, using different calibration constant sets for different systems, to give perfect descriptive power. However, such tuning may obscure the true aspects of a natural system (*2*). In addition, the effects of N abatement on many coastal areas have been quite disappointing (*3*).

There is one general dynamic phosphorus (P) model (thus far) that has yielded good predictions of phosphorus and chlorophyll in all Baltic Sea basins without basin-specific tuning and without taking N concentrations into account. The abatement strategy for the Baltic Sea should therefore focus on cost-effective P reductions, such as urban sewage treatment (*2*). Strategies designed for other estuaries, coastal areas, and lakes should also be based on methods with documented cross-systems predictive power.

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Eutrophication: Time to Adjust Expectations

D. J. CONLEY *ET AL.* ("CONTROLLING EUTROphication: Nitrogen and phosphorus," Policy Forum, 20 February, p. 1014) advocate a shift in strategies to control eutrophication of aquatic systems. We agree that the best hope for success rests with strategies couched in a systems perspective and founded on an understanding of interactions among biogeochemical cycles.

Current efforts to control eutrophication focus on repairing past damage, with systems expected to return to a desired state after obvious stressors are reduced. One approach is to restrict nutrient inputs to waterbodies that are declared impaired by limiting total maximum daily loads. Although costly, managing anthropogenic loads of macronutrients represents an essential, sensible, and feasible strategy for controlling eutrophication and reversing its effects. Increased loads of nutrients drive, support, or enable eutrophication.

Reducing loads of macronutrients, including simultaneous reductions for nitrogen and phosphorus, may not yield desired responses. Beyond lags due to "legacy loads," systems may not return to undamaged states along desired trajectories if they have entered alternative stable states or baseline conditions have shifted (*1*–*3*). For example, ocean acidification may prevent corals from regaining ascendancy on reefs, and impacts from overfishing may cascade through trophic webs to create stable but undesirable assemblages of consumers and producers. In such cases, reversing eutrophication may require restoration of habitats, repair of trophic webs, or relatively drastic projects that remove accumulated effects and thus shift systems toward previous states. In some cases, our best efforts may not produce systems that are structurally identical to a previous, desired state, so we will have to settle for restoring dynamic functions that consistently yield desired services (*1*, *2*).

Sustainable control of eutrophication in aquatic systems requires all stakeholders to acknowledge our inability to predict the exact trajectory followed by any particular ecosystem in response to management interventions, including reduced loads of nitrogen and phosphorus. Stakeholders can foster success by embracing an adaptive approach supported by monitoring that evaluates alternative actions and endpoints, promotes continual learning, and fosters progressive improvement (*4*).

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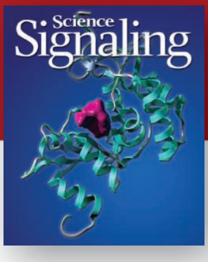
Response

WE ARE GRATIFIED THAT OUR POLICY FORUM has stimulated numerous responses on effective strategies for controlling phosphorus (P) and nitrogen (N) to reduce eutrophication in freshwater and coastal marine ecosystems.

Rigid application of P control—the only paradigm proffered by Schindler and Hecky—has been increasingly called into question even for lakes (*1*, *2*) and was rejected long ago for estuarine and coastal waters for the reasons discussed in our Policy Forum. The paradigm depends on sufficient N_2 fixation by cyanobacteria to meet the demands of algal growth that could be supported by the available P. While this is not always reached even in lakes (*1*, *2*), quantitatively significant N_2 fixation simply does not occur in the water columns of coastal ecosystems except, as we pointed out, under low-salinity conditions found in the more freshwater portions of estuaries and the Baltic Sea. The evidence cited by Schindler and Hecky for estuaries is for N_2 fixation in bottom sediments, which is seldom important in the N economy of estuarine ecosystems.

Although demonstration at the whole-ecosystem scale advocated by Schindler and Hecky is powerful, it is not usually possible to intentionally make whole-ecosystem experiments in marine systems. Nonetheless, there is substantial evidence of N limitation in coastal marine ecosystems at the whole-ecosystem and large-mesocosm scale (*3*). In fact, the example by Schindler and Hecky of P control alone succeeding in reducing eutroph-



ication in a low-salinity estuary, the Stockholm Archipelago, is only half of the story. Phytoplankton chlorophyll levels further declined after the waste treatment facilities substantially removed N (*4*).

Schelske's basic points are similar to those of Schindler and Hecky, namely, that N_2 fixation can alleviate N shortages for phytoplankton and that there have been many demonstrations of successful P-only control in lakes, such as in some of the Laurentian Great Lakes. Unfortunately, such P-only control has not been universally effective. In fact, the very reference Schelske provides to support the success of P-only control noted that it is important to consider not only P but also N loading (*5*). Shallow hyper-eutrophic lakes such as Lake Apopka frequently have blooms of cyanobacteria that do not fix N_2 and have not responded to P-load reduction, probably because of the large internal recycling of P from sediments. Schelske's calculations consider only external P loading and ignore P remobilization from sediments.

We agree with Bryhn and Håkanson that P reductions are required for improvements to be observed in the Baltic Sea; however, we differ in the need for N reductions. Their model is only for P because they believe that it is not possible to construct adequate models for nitrogen mass-balances, despite the fact that N models are common. Their model parameterization for P cycling is unusual—the boundary conditions are very different from other models used in the Baltic Sea, and their conclusions differ substantially as well (*6*–*8*), bringing into question their validity.

Jacoby and Frazer agree with us that both P and N controls should be considered, but stress that this might not yield desired responses because the damaged ecosystems may have shifted to alternative stable states. This issue fell beyond the scope of our short Policy Forum, but we have addressed it elsewhere (*9*). We certainly agree that this makes the exact recovery trajectory somewhat unpredictable and requires an adaptive approach, the first step of which is aggressive control of nutrient loads based on a rational and context-specific two-nutrient strategy.

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