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Evaluating Nitrogen and Phosphorus Control in Nutrient TMDLs

Excessive fertilization of surface waters leads to rapid algae growth. What does this mean for developing TMDLs?

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Comments

The development of total maximum daily loads (TMDLs) is causing a resurgence of interest in controlling the excessive fertilization–eutrophication–of surface waters. Nitrogen (N), phosphorus (P), and other nutrients stimulate excessive growth of algae and other aquatic plants. Of particular concern are nutrients added through the rural and urban application of inorganic and organic fertilizers. Those who fertilize lands as part of crop production, fertilize urban lawns, or dispose of waste residues (biosolids, animal manure, and compost) on land will be required as part of nutrient TMDLs to conduct comprehensive, reliable monitoring programs to ensure that the fertilizer nutrients and the constituents in the waste do not cause pollution–impairment of uses–of ground and surface waters associated with the waste management activities.

Additional Article Content

This article, based on a review by Jones-Lee and Lee (2001), provides guidance on the use of the Organization for Economic Cooperation and Development (OECD) eutrophication study results to evaluate the potential impacts of nutrient releases from agricultural and urban areas on eutrophication-related water quality in many types of water bodies. The Vollenweider-OECD eutrophication modeling approach is a powerful, reliable tool for determining the degree of nutrient control needed to achieve desired water quality for some types of water bodies. The article also provides guidance on establishing allowable nutrient loads to water bodies to protect designated beneficial uses. Particular attention is given to assessing the water-quality significance of N and P present in stormwater runoff and irrigation return waters from land areas that receive inorganic fertilizers and waste residues. Finally, the article discusses monitoring programs needed to ensure that land application of inorganic fertilizers and waste residues does not cause or contribute to impairment of the beneficial uses of surface water or groundwater.

As part of implementing the Clean Water Act requirements for TMDLs, USEPA (1999) and many states are developing control programs for aquatic-plant nutrients (N and P compounds) in wastewater discharges, irrigation return/drainage water, and stormwater runoff from urban and rural areas. Many water bodies in the

United States are listed as 303(d)-"impaired" because of excessive aquatic plant growth in surface waters; the number of listings of this type will likely increase when USEPA (1998, 2000a) and the states develop chemical-specific water-quality criteria for N and P compounds.

How Do Nutrients Affect Water Quality?

Water hyacinth

Algal bloom seen from the top of a dam, looking down at the water's surface

The nutrients N and P stimulate the growth of a variety of types of aquatic plants. When present in excessive amounts, these plants can significantly impair the beneficial uses of water bodies. Eutrophication leads to the growth of planktonic (suspended) and attached algae and can also, under certain conditions, lead to excessive amounts of higher aquatic plants, such as water weeds (e.g., water hyacinth) and others that are adverse to beneficial uses.

Lee (1971) summarized how aquatic plants can adversely impact beneficial uses. Planktonic algae and, in some cases, attached algae can cause tastes and odors in a domestic water supply. Planktonic algae can also cause shortened filter runs, increased chlorine demand, and interference with disinfection. Furthermore, as discussed by Lee and Jones (1991a), algae can, under unusual circumstances, increase the total organic carbon in the water body and thereby lead to increased trihalomethanes upon disinfection with chlorine. Another significant impact of eutrophication is an impairment of the recreational uses: contact and noncontact recreation including boating, swimming, wading, and shoreline activities.

Eutrophication of water bodies, such as Chesapeake Bay, has apparently led to the growth of certain algae (Pfiesteria) that are toxic to fish. Furthermore, in some situations, either naturally derived nutrients or those derived from cultural sources lead to red tides, in which excessive growths of certain types of algae, some of which may be toxic, occur. An area of the Gulf of Mexico is experiencing hypoxia (low dissolved oxygen), which apparently is related to the algal growth in the surface waters of the gulf. According to USEPA (2000b), the hypoxia is strongly correlated with nutrient discharges from the mouth of the Mississippi River. This river drains 40% of the lower 48 states; its watershed is home to almost a third of the US population. The Gulf of Mexico hypoxia situation has stimulated EPA and states to explore developing nutrient-control programs in the Mississippi River watershed. Increased attention will be given to the sources of nutrients in the Chesapeake Bay and Mississippi River watersheds that are causing excessive fertilization of the bay and the Gulf of Mexico.

Eutrophication of a water body can stimulate sufficient aquatic-plant growth to impair the water body's fisheries. Although the addition of nutrients stimulates overall fish production, as discussed by Lee and Jones (1991b), excessive fertilization can also significantly adversely impact the quality of fish, changing the populations from desirable game fish to rough fish such as carp. The decomposition of excessive planktonic algae can cause deoxygenation of the hypolimnion of a water body and, if severe, lead to significant dissolved-oxygen (DO) depletion in the surface waters as well. While somewhat unusual, an example of this type of situation is occurring in the San Joaquin River Deep Water Ship Channel near Stockton, CA. There, nutrients derived primarily from agricultural sources in the San Joaquin River watershed stimulate sufficient algae to lead to depletion of the oxygen resources to levels below water-quality standards throughout the water column, including surface waters (Lee and Jones-Lee, 2000a,b). This situation arises from the biochemical oxygen demand (BOD) of the algae, which exerts an oxygen demand in the water column and contributes to biotic and abiotic oxygen demand in the sediments.

The diel (24-hour) cycle of oxygen production (photosynthesis) and consumption (respiration) associated with algal growth causes increases and decreases in DO concentration over the course of a day. How low the DO concentration goes, as well as the duration of the decreases, affects how this phenomenon can impact fish. The issue that should be addressed is what it means to the aquatic-life resources of the water body to have excursions of DO below the 5-mg/l criterion for a few hours each day. These excursions can be as much as 1-2 mg/l or so below the standard during periods when there are significant algal populations in the near-surface waters.

DO depletions below 5 mg/l affect the rate of growth of fish and other aquatic life. The altered fish-growth rates are small for minor depletions below 5 mg/l; however, depletions down to 3 mg/l are acutely lethal to some fish (USEPA, 1986, 1987). An important part of developing a nutrient-control program is to clearly define the DO water-quality—standard violations that are to be controlled by nutrient management. In some instances, because of the high cost associated with meeting worst-case-based water-quality standards for DO, it might be necessary to develop a special classification of aquatic-life—related beneficial uses of water bodies. This would allow some minor impairment of the beneficial uses as a result of DO depletions below the water-quality standard associated with diel or near-sediment/water interface excursions below the standard.

Algal Nutrients and Water-Quality Problems

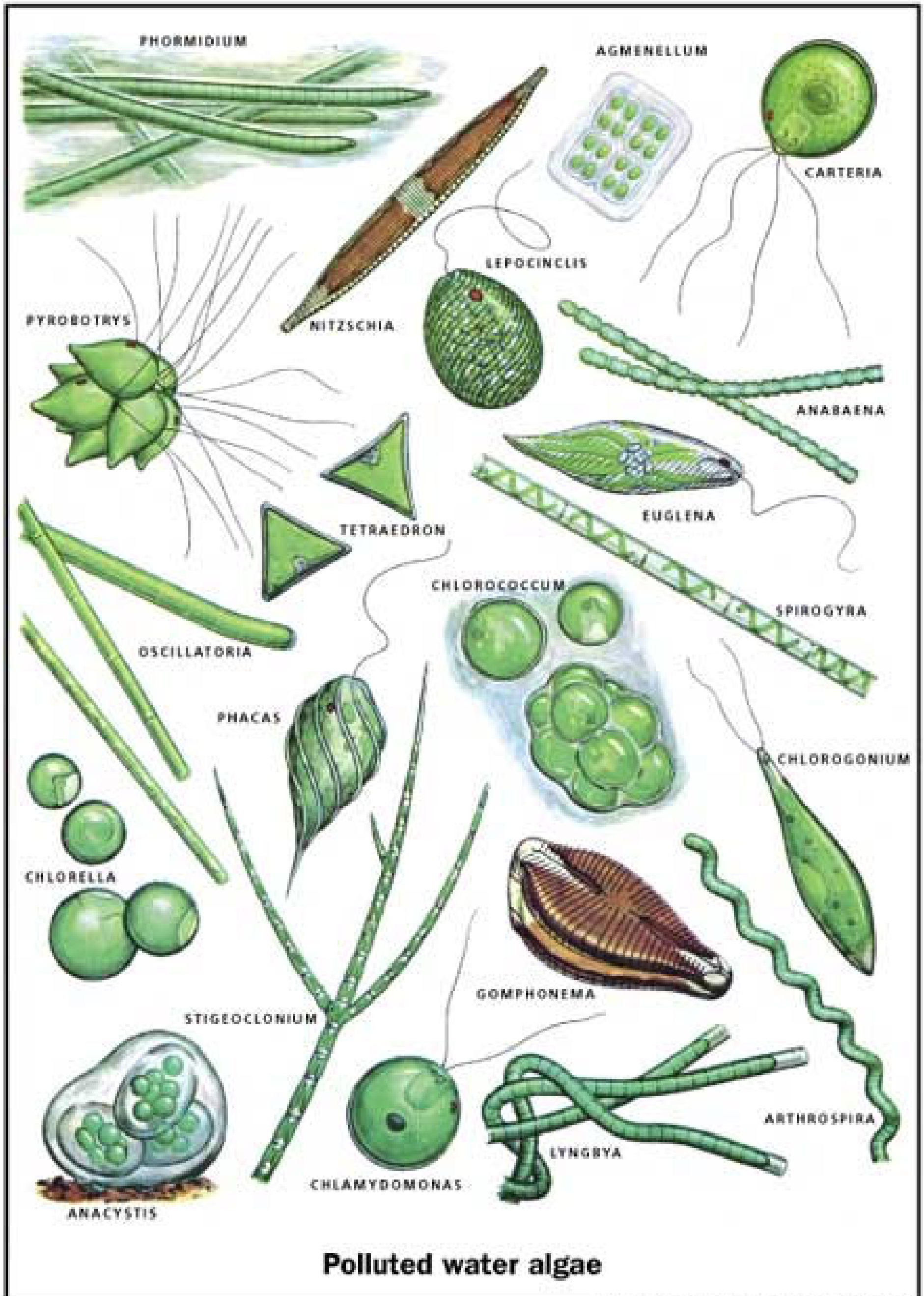

ILLUSTRATION: US PUBLIC HEALTH SERVICE

Several factors influence the relationship between algal nutrients that are added to a water body and water-quality problems. Aquatic-plant nutrients N and P exist in a number of forms; some are available and others are unavailable to support algal growth. Nitrate and ammonia are typically available forms of N. While organic nitrogen, when converted to ammonia, is available, part of the organic nitrogen is not readily convertible to ammonia through ammonification reactions. For phosphorus, it is the soluble orthophosphate that is available to support algal growth. Many forms of particulate phosphorus– for example, the phosphorus associated with inorganic particulates–are not available and typically are not readily converted to available forms. Some forms of particulate phosphorus, such as algal cells, are converted through mineralization reactions to soluble orthophosphate, which supports algal growth. Based on the review by Lee et al. (1980), for many situations associated with urban and rural/agricultural land runoff, the amount of available P in a water body can be estimated to be equal to the soluble orthophosphate plus about 20% of the particulate phosphorus.

The stoichiometry of algae is typically 106 carbon to 16 N to 1 P on an atomic basis, or 7.5 N to 1 P on a mass basis. Based on the stoichiometric composition of algae, typically either N or P is the element present in the algal environment that, when supplied at a rate less than needed, can limit the growth of algae; that is, is the limiting element. This is important in managing eutrophication-related water-quality impairment, because increasing the supply of the available limiting nutrient increases algal biomass. Similarly, reducing the amount of the available limiting nutrient can affect reduction in algal biomass. The approach for reliably determining limiting nutrients in a water body, as described by Lee and Jones-Lee (1998), involves determining the concentrations of available N and P at maximum algal biomass. If the concentrations of available N or P are greater than growth-rate-limiting concentrations under these conditions, then that nutrient is not limiting.

Algae can grow to a sufficient extent to shade themselves and, thereby, limit their further growth. Inorganic turbidity, such as is associated with erosion, and color in the water body can also reduce light penetration sufficiently to cause the algal biomass in the water body to be less than it would be if the turbidity and color were not present.

Critical Nutrient Concentrations

USEPA (1998, 2000a) is attempting to develop chemical-concentration—based, numeric water-quality criteria/standards for nutrients. EPA's proposed approach is to define a critical nutrient concentration for a particular ecoregion and type of water body; for example, a river, a lake, or an estuary. This critical nutrient concentration, which would become the state water-quality standard, would be applied to all water bodies of the particular type. In our experience, however, this approach can lead to inappropriate evaluation and regulation of critical nutrient concentrations for many water bodies. As discussed below, the approach that should be used to determine the appropriate nutrient load/concentration should be based on a site-specific evaluation considering the water body's nutrient load and its morphological and hydrological characteristics. Furthermore, the critical concentration of a nutrient is related to the response of the public in the particular area to the presence of algae or other aquatic plants.

As originally proposed, EPA's numeric nutrient criteria/standards would be applied in ways similar to the standards for heavy metals and various organics: An exceedance of the standard value would cause the water body to be considered "impaired," which in turn would lead to the development of nutrient-control programs through a TMDL. Typically, successful nutrient-control programs focus on controlling the limiting-nutrient input to the water body. It may be possible in some situations to control the particular nutrient input to the water body to make it limiting. For most fresh waters, available P is the element that is either limiting or can be made limiting for algal growth. In some areas, especially on the US West Coast and in many marine waters, N is the element that is more likely to limit algal growth. In heavily impacted areas, such as those receiving large amounts of domestic and some types of industrial wastewaters and/or agricultural runoff, neither element is limiting. Under these conditions, it is normally more cost-effective to control P inputs from wastewaters and agricultural runoff than N inputs (Lee and Jones, 1988).

Managing Excessive Fertilization

Water bodies exist in various degrees of natural fertility, from highly oligotrophic, with very few algae, to hypereutrophic, with massive naturally occurring algal blooms. The cultural activities (farming, development of municipalities, and industry) in a water body's watershed can greatly increase the flux of N and P to a water body. The issue of primary concern in managing eutrophication of a water body is the impact of cultural activities in a watershed on the increased fertility above the natural fertility that would be present in the water body. In the 1950s and 1960s, it was recognized that the excessive fertilization of water bodies was one of the major causes of water-quality-use impairment.

USEPA's National Water Quality Inventory (2000b) presents the Clean Water Act section 305(b) 1998 report to Congress on the condition of US water bodies. It shows that about 45% of lakes, reservoirs, and ponds in the states providing information for the inventory impaired water quality as a result of nutrients. Futhermore, 30% of the lakes included in the inventory were impaired because of constituents (nutrients, sediments, atmospheric deposition, and so on) derived from agricultural sources. About 12% of the impaired lakes were impaired because of constituents in wastewater sources discharged in them. As indicated in the National Water Quality Inventory, excessive fertilization of water bodies is one of the most significant causes of water-quality impairment in the US.

Substantial literature was developed in the 1960s and 1970s devoted to managing eutrophication of water bodies. It was established, primarily by Vollenweider (1975, 1976), that the impact of nutrients on a water body (a lake or a reservoir) is dependent not only on the water body's nutrient loads and concentrations but also on its morphology (mean depth) and hydrology (hydraulic residence time). This situation makes EPA's attempts to develop standardized, ecoregionwide critical loads/concentrations of nutrients to water bodies questionable. Although a variety of modeling approaches (Ambrose et al., 1988, 1993a,b; Bowie et al., 1985; USEPA, 1997) can be used to attempt to relate nutrient loads to a water body to the water body's planktonic algal growth, the most comprehensive and reliable approach is the result of the Vollenweider-OECD eutrophication study, a 22-country, 200-water-body, $50 million effort that took place over a five-year period in the 1970s in Western Europe, North America, Japan, and Australia (OECD, 1982; Vollenweider and Kerekes, 1980). Figure 1 presents the relationships developed from the Vollenweider-OECD studies for the US water bodies (Rast and Lee, 1978; Lee et al., 1978).

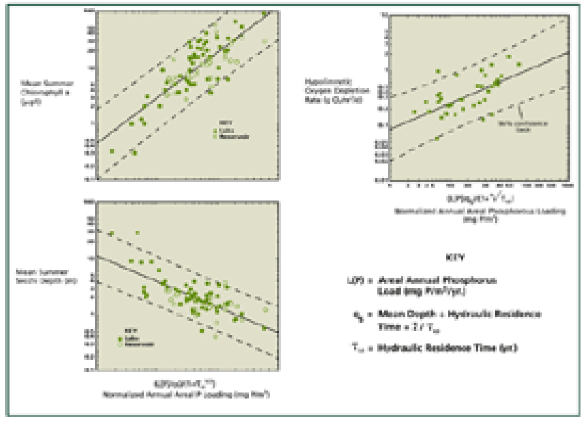

Figure 1.

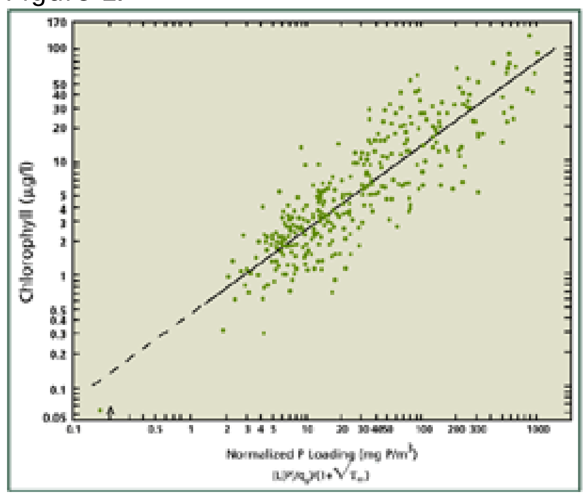

Figure 2.

Figure 2 presents the database that exists now, which is in excess of 750 water bodies located in various parts of the world (Jones and Lee 1982, 1986). The abscissa in Figures 1 and 2 is Vollenweider's normalized phosphorus load. This normalized load to a lake or a reservoir is approximately equal to the average annual concentration of P in the water body. As shown in these figures, there is remarkably good agreement between the normalized P load (in-lake phosphorus concentration) and the average planktonic algal chlorophyll. Each of the dots shown in Figure 2 represents a water body that has been evaluated by a nutrient load—eutrophication response study conducted over at least one year. A wide range of types of water bodies is represented, including a permanently ice-covered Antarctic lake (Jones-Lee and Lee, 1993), shallow farm ponds, and Lake Superior in the US-Canadian Great Lakes. They include several estuarine and near-shore marine waters as well (Lee and Jones, 1981, 1989).

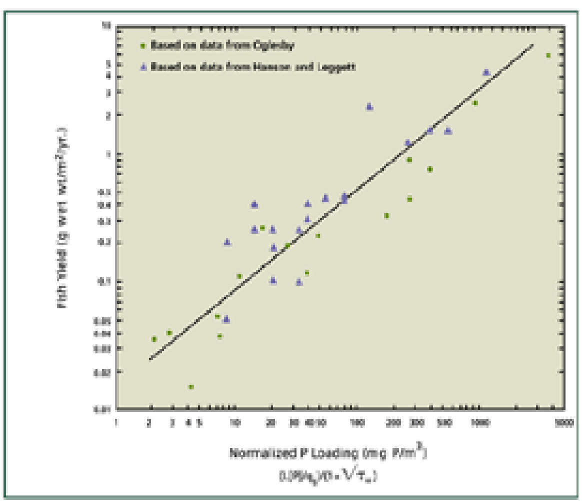

Figure 3.

Lee and Jones (1991b) compiled information on the relationship between P loads to water bodies and their yields of fish (Figure 3). This figure shows that there is a good relationship between the normalized P load and the fish biomass present in a water body. As the nutrient loads increase, there is a shift in the type of fish present. In temperate climates, nutrient-poor water bodies that stratify and thereby maintain cooler waters in the summer in the hypolimnion can maintain cold-water fisheries. As the fertility of the water body increases, however, deoxygenation of the hypolimnion leads to sufficient DO depletion to prevent cold-water fish from oversummering in that area.

As discussed by Jones and Lee (1988), the relationships shown in Figures 1, 2, and 3 can be used to evaluate overall ecosystem functioning with respect to relating nutrient loads to primary and secondary production. Of particular concern is whether there are toxicants in the water body that are inhibiting aquatic-organism production at various trophic levels. With few exceptions, the relationships shown in Figure 2 are applicable to a wide variety of water bodies (principally lakes and impoundments) located throughout the world.

Based on the normalized P load to water bodies, it is possible to predict–with sufficient reliability for management purposes–the planktonic algal chlorophyll that should be present in the water body and, most importantly, how changing the P load to the water body will change the overall planktonic algal chlorophyll content of the water. As discussed by Rast et al. (1983), the relationships shown in Figures 1 and 2 can be used to evaluate the impact of adding or removing P from a particular source on the overall fertility of the water body.

Recommended Evaluation Approach

Some people espouse, as part of eutrophication management programs, that "every little bit" of nutrient control helps. This claim, however, is not supported by knowledge of how nutrients impact eutrophication-related water-quality problems. Lee and Jones (1986) found that at least a 20-25% reduction in nutrient load to a water body is needed to produce a discernible improvement in water quality. Except for situations in which a major point source, such as a domestic wastewater discharge, is the primary source of nutrients for a water body, eutrophication management programs should be based on watershed approaches in which all sources of available forms of nutrients are evaluated with respect to their effect on the excessive fertilization of the water body in question. The components of a watershed-based eutrophication management program are summarized below.

Defining the Eutrophication-Related Problem

It is extremely important to properly define the water-quality problem(s) caused by excessive nutrient input to a water body of concern. As part of defining the problem, stakeholders should develop consensus on the desired degree of control of the use impairment. For domestic water supplies, this would likely focus on the frequency of noxious algal blooms that cause tastes and odors or shortened filter runs. For recreational use of waters, water clarity controlled by planktonic algae (Secchi depth) is a useful measure.

Lee et al. (1995 a,b,c) discuss an approach for establishing the desired trophic state (degree of fertility) for a water body, with particular attention to water clarity (Secchi depth) as a quantitative indicator of the aesthetic quality of the water. As shown in Figure 1, there is an inverse relationship between Secchi depth and planktonic algal chlorophyll in water bodies in which Secchi depth is controlled by planktonic algae.

Low DO concentrations in the hypolimnion (hypoxia) are an important consequence of eutrophication of water bodies that density stratify as a result of temperature and/or salts. Figure 1 shows the initial results developed by Rast and Lee (1978) relating the normalized P load to a water body to the rate of hypolimnetic oxygen depletion. Depletion of DO in the hypolimnion of a water body is controlled by the BOD of the algae raining down through the thermocline and the oxygen demand of the sediments, as well as by the volume of the hypolimnion. Fitzgerald (1964) measured the oxygen demand of algae, which on a mass basis is similar to that of domestic wastewaters. The oxygen demand of sediments is normally caused primarily by abiotic reactions involving ferrous iron and sulfides. It is possible, through relatively simple models, to relate these two sources of oxygen demand to the amount of planktonic algal chlorophyll that develops in the surface waters of a water body. As a result, it is possible to relate the nutrient loads to a water body to its hypolimnetic oxygen-depletion rate. The impact of the hypolimnetic oxygen-depletion rate on the water quality–i.e., on DO concentration–depends on the volume of the hypolimnion, which is a function of the water body's morphology.

Lee (1970) reviewed the factors influencing the exchange of oxygen demand and other constituents associated with water-body sediments, pointing out that the primary mechanism for this exchange is the degree of mixing of the sediments into the overlying water column. This enables the constituents, including the oxygen demand associated with the interstitial waters of the sediments, to be mixed into the water-column waters near the sediment/water interface. This mixing process is controlled primarily by physical processes, such as currents induced by water movement near the sediment/water interface. Also, biological processes, such as anaerobic production of methane and carbon dioxide, can be a mechanism for stirring of sediments. Aquatic-organism foraging and bioturbation are also important mechanisms for mixing of sediments, thereby promoting sediment/water exchange reactions.

It is important to understand that the Vollenweider-OECD and other eutrophication-modeling efforts are applicable only to planktonic algae. They do not reliably predict how nutrients impact attached algae or attached or floating water weeds and macrophytes such as water hyacinth.

Assessing Limiting Nutrients and Their Sources

As part of its efforts to develop water-quality criteria for nutrients, USEPA (2000a) is attempting to use total P, rather than algal-available P, as the basis for nutrient criteria/standards. This approach can readily lead to significant errors in relating P loads to water bodies to the planktonic algal chlorophyll that can develop from the P load. While the relationships shown in Figures 1 and 2 are based on a normalized total P load, the data shown in these figures are for water bodies in which most of the added P was in an algal-available form (i.e., soluble ortho-P or particulate phosphorus that can readily be converted to soluble ortho-P). As discussed above, only about 20% of the particulate P in land runoff from agricultural or urban areas can be expected to be converted to algal-available P (Lee et al., 1980).

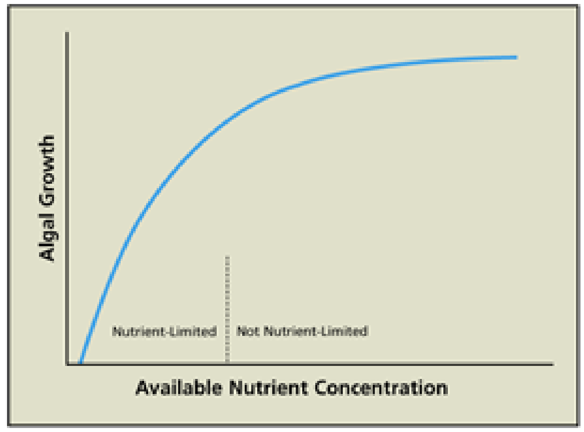

Figure 4.

In some water bodies, most added P is associated with inorganic particles because of high rates of erosion in the watershed. Under these conditions, the algal-available P load should be initially estimated based on the general finding that soluble ortho-P plus 20% of the particulate phosphate load is available to support algal growth. Site-specific investigations incorporating algal assay procedures described by Lee et al. (1980) can be used to evaluate available P loads to a particular water body.

With respect to estimating the amount of organic nitrogen that converts to ammonia/nitrate, organic nitrogen, primarily in the form of algal cells or other recently developed organic-N, will essentially completely convert to algal-available forms over a several-week period. Aged organic-N can be highly refractory, however, where only a limited amount will convert to algal-available N.

In developing a eutrophication management program, it is important to assess whether N or P is limiting or could be made to limit algal growth that impacts the water-quality problem of concern. If neither is limiting–i.e., both remain in surplus at peak biomass–then far greater control of nutrient sources will be needed to achieve growth-rate-limiting concentrations and sufficient reduction beyond that to effect change in eutrophication-related beneficial uses. Figure 4 is a diagrammatic representation of this situation. Typically, excessively fertile water bodies have nutrient concentrations on the growth-rate plateau area in Figure 4. This necessitates a significantly larger, more expansive nutrient-control program than if the excessive-fertilization water-quality problems occurred in the region of Figure 4 where the growth rate of algae is proportional to nutrient concentrations. Although it is relatively easy to control 90-95% of the P in typical secondary domestic wastewater discharges through additional wastewater treatment, from the information available in the Chesapeake Bay watershed and the lower Great Lakes watersheds, it appears that it will be extremely difficult to control N and P from land runoff more than about 40% (Sharpley, 2000; Logan, 2000). Lee and Jones-Lee (2001) have recently presented a discussion of the approach that should be followed in developing nutrient-control programs in runoff from agricultural lands. These same approaches are applicable to nutrient control associated with runoff from urban areas.

An important part of developing a technically valid, cost-effective nutrient-control program is understanding of the hydrology and physical limnology/oceanography of the water body of concern–assessing whether added nutrients are mixed throughout the water body or are present in some areas in higher concentrations than in others. Two types of situations are of primary concern. One is where the inlet to the water body is near the outlet. This can lead to short-circuiting of nutrients through the water body without the opportunity for phytoplankton to fully develop on the added nutrients. The other is long, thin water bodies in which nutrients added at one end are not rapidly mixed throughout the water body. Under these conditions, it is appropriate to divide the water body into plug-flow segments representing about one to two weeks' travel time per segment, then determine the amount of algal growth that occurs in each segment. The available nutrients removed through algal growth are used to correct the total nutrient load to the water body, where this corrected load is that which is to be used in the next plug-flow segment.

An assessment should be made of the water body's hydraulic residence time (volume divided by annual tributary inflow and precipitation). If it is less than a year, nutrients added during the high-flow winter/spring period may be flushed through the water body and therefore might not be available to support the algal growth during the following summer. Under these conditions, the nutrient load should be assessed on a monthly basis; the nutrients that contribute to the summer/fall algal growth are the nutrient loads that determine the phytoplankton levels reached during the summer.

The Vollenweider-OECD eutrophication modeling approach is not applicable to water bodies with very short residence times. The algae in water bodies with summer hydraulic residence times of less than one to two weeks does not develop to the maximum extent possible based on the nutrient loads to the water body. These types of water bodies would have lower chlorophyll than expected based on its annual normalized P loads (Figure 2).

Developing a Nutrient Load—Eutrophication Response Relationship

A key component of any nutrient-control—based program for excessive-fertilization management is the development of a nutrient load—eutrophication response relationship for the water body of concern. Because of the simplicity of its use and demonstrated reliability for a wide variety of water-body types located throughout the world, the Vollenweider-OECD eutrophication modeling approach should be evaluated as a potential tool for relating nutrient loads to a water body to the planktonic algal growth that develops in the water body. Using the guidance provided here and by Jones and Lee (1982, 1986), the normalized P load to the water body should be evaluated. This load should be used to predict, through Figures 1 and 2, the average planktonic algal chlorophyll that would be expected to be present during the summer. Jones et al. (1979) found that the maximum summer chlorophyll that develops in water bodies is about 1.7 times the mean summer chlorophyll.

If the planktonic algal chlorophyll found in the water body is within the range of values shown in Figure 2, then there is reasonable certainty that the Vollenweider-OECD eutrophication modeling approach, using the database developed by Jones and Lee (1982, 1986), can be used to predict the amount of P load reduction that needs to be achieved in order to affect the desired water-body water quality as measured by planktonic algal chlorophyll. If, however, there is poor agreement between the measured planktonic algal chlorophyll and the values predicted from the Figure 2 relationship, then either the database upon which the evaluation is being made for the particular water body is unreliable or the application of the Vollenweider-OECD eutrophication modeling approach is being done incorrectly (e.g., availability of P load or hydrological and morphological characteristics are not properly addressed).

Implementing Nutrient-Control Programs

Once the desired level of N-P control has been established, then an evaluation needs to be made of the relative significance of various sources of the available nutrient or nutrients that need to be controlled. At this point, an evaluation can be made of the potential role of the water-quality significance of any fertilizer (inorganic or organic), including manure and biosolids-derived nutrients, that leads to the eutrophication of a water body. It is likely, based on current approaches for nutrient-based TMDL development, that all nonpoint sources of nutrients will be required to reduce their nutrient export by a specified percentage of the total export from the area.

Expected Rates of Recovery

One of the issues of primary concern in a eutrophication management program based on tributary nutrient-input reduction is whether the nutrients in the sediments of the water body will represent a significant source of N and/or P so as to greatly inhibit the rate of recovery. Sediments of a eutrophic water body often contain large amounts of N and P in particulate forms. As a result, nutrients in the sediments could maintain high levels of algae even though nutrients derived from the watershed have been significantly reduced.

Sonzogni et al. (1976) investigated the rate of recovery of lakes upon reduction of the nutrient input and found that a P residence time best described this rate. The P residence time of a water body is analogous to the hydraulic residence time and is determined by assessing the total mass of P in the water column divided by the annual P load. The P residence time of a water body typically is much shorter than the hydraulic residence time. For example, for Lake Michigan, the hydraulic residence time is about 100 years, while the P residence time is about six years. The rate of recovery of a water body upon altering P loads can be estimated as being equal to three times the P residence time. Sonzogni et al. found that this approach effectively tracked the rate of recovery for several water bodies that had experienced reductions in P input from the watersheds.

The release of P from sediments is incorporated into the phosphorus residence-time model because it's included in the total mass of P in a water body's water column. The fact that water bodies recover in accord with P residence-time models demonstrates that the primary source of nutrients for many water bodies is the watershed and that while the sediments are a source of nutrients, they are not the dominant source controlling the eutrophication of water bodies. For situations in which N is the controlling element governing phytoplankton development, N residence time can also be determined.

Phased-Nutrient TMDL

The initial modeling of eutrophication response to nutrient loads as part of a TMDL nutrient-control program is designed to develop the necessary information to formulate a Phase I estimate of the amount of nutrient control needed to achieve a desired eutrophication-related water quality. If the nutrient-control program is part of a TMDL, in addition to reducing the nutrient inputs to achieve the desired water quality, it will be necessary to further reduce the nutrient loads to achieve a TMDL-required safety factor. It should be understood that, even with the OECD eutrophication study's modeling approach, it is necessary to specifically tune the nutrient load—eutrophication response relationships to a particular water body.

EPA has adopted a phased approach for TMDL implementation, where the Phase I constituent load is understood to be a rudimentary estimate of the load reductions that will be needed to achieve the desired water quality. The Phase I estimate of loads, coupled with the safety factor, provides a starting point for meaningful modeling load-response relationships. Phase I of the load-response relationship provides the opportunity to develop the water body—specific data for loads and responses to greatly improve the load-response modeling for that water body.

Phase II should be developed if the desired water quality is not achieved in Phase I. Because of year-to-year climate-related variability in nutrient load—eutrophication response relationships for water bodies, it is recommended that at least three, and preferably five, years of Phase I monitoring be conducted after the nutrient-load reductions have been implemented and the water body has adjusted to the new nutrient loads. This adjustment period is estimated as three times the limiting-nutrient residence time for the water body. Therefore, since it is likely to take several years after a TMDL load allocation has been developed before the nutrient-control programs are in place and operating reliably, Phase II may not be implemented until at least five, and sometimes 10 or more, years after the TMDL and its allocations have been developed or agreed to. Because it is possible, in many TMDL situations, that there will be litigation involving stakeholders objecting to the load-reduction allocation provided them, this could delay the initiation of Phase II from five to 15 years after the TMDL is first formulated and allocated.

Nutrient-Load Reduction Allocation

The allocation of allowable nutrient loads among the dischargers can be assigned based on respective responsibilities, where each discharger is required to reduce its load by a certain amount of its share of the total load. However, economic, political, and legal issues can often influence the allowed nutrient discharges from the various sources. Basically, nutrient-load reduction allocation is a societal process whereby society can decide which of the dischargers of nutrients, or others who influence how the nutrients impact water quality, should financially support nutrient control. For example, it might be appropriate to require domestic wastewater treatment plants to remove 90-95% of the P, which can typically be done at a cost of a few cents per person per day for the population served by the treatment plant, and to require 30-50% control of P from agricultural land runoff/discharges.

Another area receiving increasing attention is pollutant-load trading, whereby one type of discharger will remove a greater percentage of nutrients than other dischargers. Rather than spending large amounts of money to remove the last percent or so of a nutrient from the wastewater discharges, a municipality might find that it is more cost-effective to help fund agricultural interests to remove nutrients.

It is extremely important that pollutant-trading programs be based on available forms of nutrients and not on total nutrients, which include large amounts of unavailable nutrients. Numerous technically invalid pollutant-trading programs have been developed, which incorporate trades of discharge of unavailable nutrients for discharge of available nutrients. Another aspect of pollutant trading is the examination of both near-field impacts of nutrients (i.e., near the point of discharge) and far-field (i.e., downgradient or water body—wide) effects. Nutrients discharged into bays or the arms of a larger water body can readily have a significant adverse effect in the bay or arms, but not in the water body overall.

Conclusions

All sources of N and P discharged to water bodies experiencing excessive fertilization–which include many inland water bodies and much of the US East Coast, Gulf Coast, and some West Coast near-shore marine waters–will be subject to increased scrutiny as part of the TMDL efforts. All sources of nutrients will likely receive increased scrutiny as contributors to eutrophication of water bodies. It will be important for all managers of sources of these nutrients to develop and actively participate in monitoring programs to define on a water body— or watershed-specific basis the role that the source plays. It will also be important to reliably evaluate the impact on a water body of eutrophication-related water-quality problems associated with types of nutrients and with particular nutrient sources and how reducing those nutrient sources impacts those problems.

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