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September 3, 2009

Dr. Thomas Armitage
Designated Federal Officer (DFO)
EPA Science Advisory Board (1400F)
U.S. Environmental Protection Agency
1200 Pennsylvania Avenue, NW.
Washington, DC 20460

RE: SAB Review of Nutrient Criteria Guidance

Dear Dr. Armitage:

This letter is written to provide technical input to your upcoming review of Nutrient Criteria Guidance. Since 1974 when I first began studying the impact of phosphorus on Great Lakes water quality, I have devoted a great part of my scholarship to eutrophication. Although I initially focused on lakes and reservoirs, I shifted my emphasis to streams when I joined the faculty at the University of Colorado in 1986. It was there that I first recognized the significant and sometimes subtle differences between lakes and flowing systems like rivers and streams. Among other things, this resulted in my developing EPA-sponsored software that is expressly designed to simulate stream eutrophication (Chapra et al 2008).

As I understand it, the proposed approach seeks to directly correlate total phosphorus (TP) concentration with the health of invertebrate populations. Thus, phosphorus is treated as if it were a toxic substance that directly interferes with the viability and functioning of the biota.

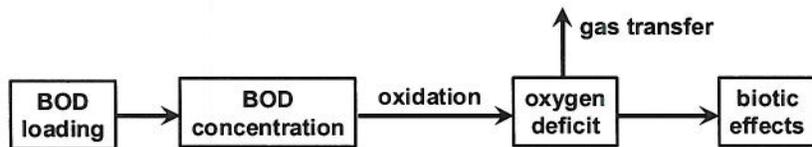
Because this approach is based on a flawed conception of the connection between nutrients and ecosystem health, I believe that its adoption would represent a grave mistake. Beyond being vulnerable to legal challenge, I am much more concerned that its adoption would ultimately be ineffective. That is, it could lead to costly controls that would not protect our precious stream ecosystems.

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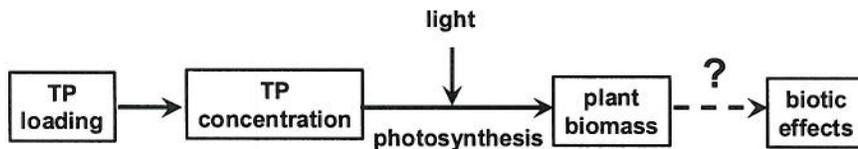
An analogy with one of the earliest water-quality management problems is instructive in illustrating my concern. In the early 20th century, point discharges of untreated urban sewage resulted in low oxygen concentrations in many of our nation's rivers and estuaries. Because adequate dissolved oxygen (DO) levels are necessary for most forms of aquatic life, the stress on ecosystems was great. Aside from direct impacts on fish and macroinvertebrates, low oxygen also triggers a number of secondary effects such as the generation of noxious odors.

Water-quality engineers recognized early on that the root cause of oxygen depletion was the presence of organic carbon and reduced nitrogen compounds in the urban wastewater. In the simplest sense, these compounds were collectively called biochemical oxygen demand or BOD. Once introduced into the river, the BOD was broken down by bacteria. Because the bacteria consumed oxygen to effect the break down, the river's oxygen resources were depleted.

As depicted in Figure 1a, the cause-effect sequence for this problem is BOD loading → BOD concentration → dissolved oxygen concentration deficit¹ → biotic impact. Notice that along with BOD oxidation, the deficit is also determined by gas transfer or reaeration.



(a) The BOD/Dissolved Oxygen Problem



(b) The TP/Biomass (AKA Eutrophication) Problem

Figure 1 Analogy between two river water-quality problems: (a) dissolved oxygen and (b) eutrophication. The question mark on the link between plant biomass and biotic effects is not meant to suggest that there is no connection. Rather, it is meant to indicate the complexity of this connection.

Although this is a very simplified representation of the problem, the important point is that the direct cause of the ecosystem impairment is the oxygen deficit, not the BOD concentration. Hence, although we might ultimately alleviate the ecosystem stress by reducing the BOD loading, the actual BOD concentration would in fact not be directly correlated with the low oxygen levels.

This can be clearly illustrated by using the classic Streeter-Phelps model to develop a plot of BOD and oxygen in a river below a single point source of BOD. Although this model is simple, it accurately captures the interplay between BOD and oxygen in one-dimensional rivers².

¹ The deficit measures the difference between the saturation concentration and the actual DO level (Figure 2). A high deficit, therefore, indicates low oxygen and a high ecosystem stress.

² In this context, one-dimensional means that changes only occur longitudinally and that the stream is well-mixed laterally (bank-to-bank) and vertically (with depth).

As in Figure 2, the BOD is highest at the discharge point ($x = 0$) where the BOD loading is introduced. It then decreases downstream as the BOD is broken down via oxidation. The BOD oxidation in turn consumes oxygen which leads to a rapid decrease in oxygen concentration. However, as the DO falls, oxygen transfer across the air-water interface increases. Eventually the oxygen profile levels off when the oxidation loss is balanced by reaeration. Thereafter, oxygen levels begin to climb as reaeration becomes dominant and the stream recovers.

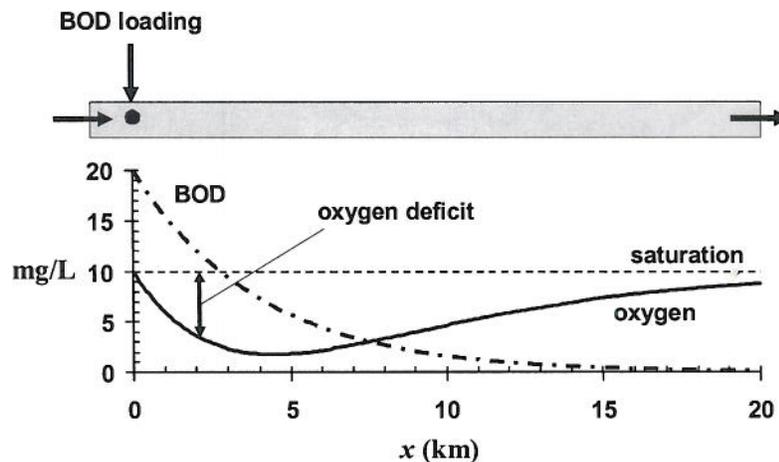


Figure 2 Simulation of BOD and dissolved oxygen versus distance below a single point source of BOD into a one-dimensional river.

The most important feature of Figure 2 is that there is absolutely no spatial correlation between the in-stream BOD and oxygen levels. For example the highest BOD concentration and the lowest deficit both occur at the mixing zone ($x = 0$). Although BOD certainly causes oxygen depletion, it would therefore be ludicrous to specify an instream BOD concentration criterion in order to ensure an adequate DO level. Instead, the correct approach is to set an oxygen criterion that is directly connected with ecosystem health. A model can then be employed to link the oxygen concentration back to the BOD loading. This, of course, is how oxygen has been so effectively managed over the past century.

Beyond illustrating how river BOD and DO evolve and interact, employing a cause-effect model yields additional benefits. For example, rather than reducing the BOD load, the analysis suggests that an alternative means to raise oxygen levels might involve enhancing gas transfer (e.g., oxygen diffusers, artificial waterfalls, etc.). Thus, the scientific approach reveals possible alternative remediation strategies that in certain cases might actually be more cost effective than source controls.

So how is this example relevant to the stream eutrophication problem? As illustrated in Figure 1b, a very simple representation of stream eutrophication is that nutrient loading (in this example, phosphorus³) results in increased stream nutrient concentration which in turn leads to increased plant biomass. Thus, as BOD is to oxygen deficit, phosphorus is to plant biomass. And just as it would be incorrect to specify an instream BOD concentration criterion to solve the

³ Notice that in Figure 1, we have assumed that phosphorus is the limiting nutrient. It should be understood that this is an assumption and that another nutrient (e.g., nitrogen) or light might in fact be limiting.

oxygen problem, it is equally misguided to specify an instream total phosphorus concentration criterion to solve the eutrophication problem.

Further, the subsequent connection between increased biomass and biotic impacts is not as well understood as for oxygen. In fact, one such connection involves the impact of excessive plant growth on stream oxygen via both direct (photosynthetic gains and respiration losses) and indirect (when plants die they become BOD) pathways. However, the deleterious effects undoubtedly involve other non-oxygen related factors such as direct habitat impairment and shifts to undesirable plant species.

As with the previous oxygen example, beyond illustrating how river nutrients and biomass evolve and interact, the cause-effect model can also suggest alternative remediation approaches. For example, rather than reducing the nutrient load, Figure 1*b* indicates that an alternative means to reduce biomass might involve decreasing solar radiation. For example, this could be accomplished by planting riparian vegetation to create a canopy over the stream. As with oxygen, there might be cases where such alternatives could prove useful and cost-effective.

As a final technical note, it is important to distinguish between floating (i.e., phytoplankton) and attached (i.e., periphyton, filamentous algae and macrophytes) plants when dealing with stream eutrophication. For phytoplankton, which tend to dominate in deeper rivers, the direct analogy expressed in Figure 1 holds (see App. 1 for a detailed analysis). That is, it is incorrect to set a TP criterion in order to manage river phytoplankton biomass.

For attached plants, the situation is much more complex. For systems dominated by macrophytes, nutrient management must consider whether the plants can draw nutrients from the sediments via their roots. For such situations, managing instream TP concentration would be counterproductive as photosynthesis would effectively be independent of water nutrients.

For filamentous algae (e.g., *Cladophora*) which draw nutrients directly from the water, there is certainly a closer connection. However, it is well known that (a) the photosynthesis rate of such organisms depends on their internal nutrient levels and (b) they only take up dissolved inorganic nutrients from the water. Hence, regulation based on water TP concentration would be ill-founded.

The same point can be made for periphyton but with additional nuances. Whereas filamentous algae extend up into the water column, periphyton exist as biofilms on substrates such as bottom rocks. In such cases, their nutrient uptake can be influenced by transport limitations on the delivery of dissolved inorganic nutrients from the water into the biofilm. Such limitation would be dependent on stream hydraulics and hence related to factors such as stream velocity, depth, etc. Further, because they are bottom dwellers, their growth would obviously be highly dependent on the delivery of light to the stream bottom. Hence, along with nutrients, the clarity and depth of the overlying water would have to be considered in determining their biomass. Consequently, a direct correlation with water TP concentration would seem to be overly simplistic.

In conclusion, I hope that the foregoing provides some indication of my great concern over the issue of stream eutrophication management. As a concerned environmentalist, as well as a lifelong fly fisherman, I truly support effective regulations to protect our nation's great rivers and streams. However, as an environmental scientist and engineer, I also know that without a sound scientific basis, such regulations are likely to fail.

Please let me know if I can provide any further information.

Sincerely,

A handwritten signature in blue ink, appearing to read "Steven C. Chapra".

Steven C. Chapra, Ph.D.

APPENDIX 1.

Why TP Concentration Standards are Inappropriate for Managing Phytoplankton Biomass in Rivers

This appendix addresses the validity of a total phosphorus concentration standard as an effective approach for managing eutrophication of rivers and streams dominated by phytoplankton. In brief, I believe that the idea of river total phosphorus criteria originates from the misguided notion that effective lake management approaches can be seamlessly (and thoughtlessly) transferred to rivers and streams.

In the late 1960's and early 1970's, several limnologists suggested that total phosphorus concentration could serve as an effective trophic state indicator (e.g., Vollenweider 1968, 1969, 1975; Dillon and Rigler 1975). In particular, Richard Vollenweider posited that lakes with total phosphorus concentrations less than 10 µgP/L would tend to be oligotrophic whereas those with greater than 20 µgP/L would tend to be eutrophic.

Although Vollenweider himself repeatedly stated that these were approximate guidelines and not hard thresholds, the values were adopted by many lake managers as quantitative goals for managing lake eutrophication. And in fact, the approach has been a useful component of nutrient remediation schemes for a number of important systems including the Laurentian Great Lakes.

So why might the approach work for lakes and not for streams? The answer to this question lies in fundamental differences between these two types of natural waters.

Simple Lake Models

In effect, the viability of the Vollenweider approach is predicated on the functioning of the particular lakes he studied; that is, deep, stratified, phosphorus-limited lakes in temperate regions with long residence times (i.e., greater than a year). In such lakes, Vollenweider (and others) assumed that the spring total phosphorus concentration was a prime determinant of plant production over the ensuing summer growing season.

For this assumption to strictly hold, once the lake stratifies in late spring, the epilimnion must essentially behave as a batch or closed system. Thus, plant growth over the ensuing summer is primarily dictated by the finite store of nutrient represented by the spring phosphorus concentration rather than by external loads. The average summer level of biomass is then determined by the recycle of this pool between inorganic and organic forms. Empirical support for the approach was provided by a number of empirical correlations. The chief examples of these were logarithmic plots suggesting strong correlations between summer average chlorophyll *a* concentrations and spring total phosphorus concentration (e.g., Sakamoto 1966, Dillon and Rigler 1974, Oglesby and Schaffner 1975).

A simple computation can be used to illustrate how such an approach is viable for lakes. First, as originally suggested by Schnoor and O'Connor (1980), total phosphorus can be divided into three components

$$TP = p_p + p_i + p_o \tag{1}$$

where p_p = phytoplankton phosphorus ($\mu\text{gP/L}$), p_i = inorganic phosphorus ($\mu\text{gP/L}$), and p_o = non-phytoplankton organic phosphorus ($\mu\text{gP/L}$). If the chlorophyll *a*-to-phosphorus ratio is assumed to be $1 \mu\text{gChla}/\mu\text{gP}$ (Reynolds 1984), this means that p_p can be directly interpreted as a measure of phytoplankton biomass.

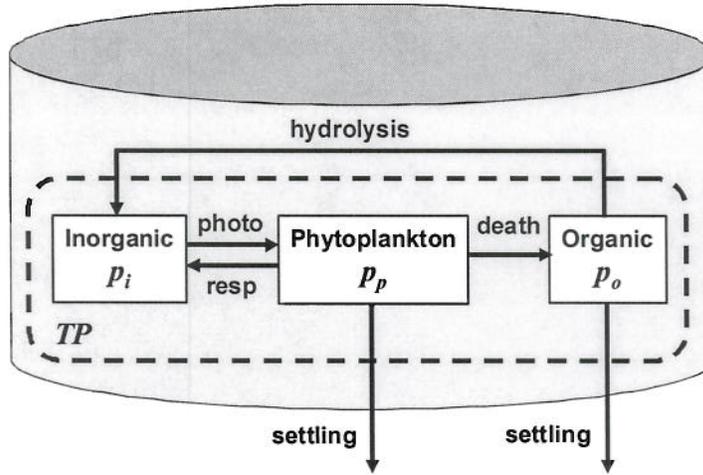


Figure 3 The epilimnion of a stratified lake as a closed or batch system with total phosphorus divided into phytoplankton, inorganic and organic phosphorus.

If the lake's epilimnion is idealized as a closed system or batch reactor (Figure 3), the following mass-balances can be written for each phosphorus species

$$\frac{dp_p}{dt} = k_g \frac{p_i}{k_{sp} + p_i} p_p - k_r p_p - k_d p_p - k_s p_p \quad (2)$$

$$\frac{dp_i}{dt} = -k_g \frac{p_i}{k_{sp} + p_i} p_p + k_r p_p + k_h p_o \quad (3)$$

$$\frac{dp_o}{dt} = k_d p_p - k_h p_o \quad (4)$$

where t = time (d), k_g = maximum growth rate at constant light and temperature (/d), k_{sp} = phosphorus half-saturation constant ($\mu\text{gP/L}$), k_r = respiration/excretion rate (/d), k_d = death rate (/d), k_s = settling rate (/d), and k_h = hydrolysis rate (/d).

Given reasonable values for the parameters and a set of initial conditions (Table 1), these equations can be integrated numerically to simulate how the various phosphorus species change with time. For the present example, the phytoplankton settling velocity is assumed to be negligible and the initial *TP* concentration is set at a solidly eutrophic level (for a lake) of $40 \mu\text{gP/L}$.

Table 1 Parameters and initial conditions used to simulate phytoplankton and phosphorus concentrations for the epilimnion of a stratified lake during the summer growing period.

Parameter	Value	Units
k_g	0.5	d^{-1}
k_{sp}	2	$\mu gP L^{-1}$
k_r	0.2	d^{-1}
k_d	0.1	d^{-1}
k_s	0	d^{-1}
k_h	0.05	d^{-1}
Initial conditions:		
p_p	1	$\mu gP L^{-1}$
p_i	38	$\mu gP L^{-1}$
p_o	1	$\mu gP L^{-1}$

Before examining the solution, it is important to recognize that the initial TP concentration corresponds to the initial mass of total phosphorus in the epilimnion normalized to the epilimnetic volume,

$$TP_0 = \frac{M_{TP}}{V_e} \quad (5)$$

where TP_0 = the initial TP concentration ($\mu gP/L \equiv mgP/m^3$), M_{TP} = initial mass of TP (mgP) and V_e = epilimnion volume (m^3). Because the epilimnion is treated as a batch (i.e., closed) system, the solution therefore describes how this initial mass is distributed among the three forms of phosphorus as time unfolds.

The results are displayed in Figure 4. Because the inorganic P concentration is well above the half-saturation constant, the phytoplankton initially grow rapidly as inorganic phosphorus is efficiently converted to phytoplankton biomass. Growth continues until the inorganic phosphorus level approaches the half saturation constant whereupon a peak is reached. At this point, growth has become sufficiently limited that it is exactly balanced by respiration and death losses. Thereafter, the phytoplankton levels decline until the solution approaches a stable steady state. This asymptote represents the condition when phytoplankton growth exactly balances phosphorus recycle. During this whole evolution, organic phosphorus builds slowly due to phytoplankton death until it also approaches an asymptote representing a balance between death gains and hydrolysis losses.

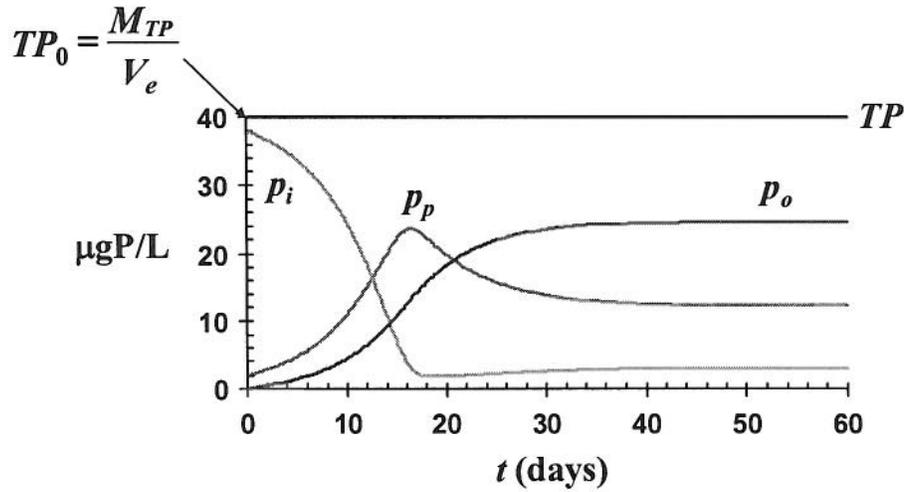


Figure 4 The temporal evolution of phytoplankton and phosphorus species in the epilimnion of a stratified lake during the summer stratified period.

Because of the assumption of zero settling, the total P concentration is constant. This allows the component concentrations at the stable steady state to be computed exactly as

$$p_i = \frac{k_r + k_d}{k_g - (k_r + k_d)} k_{sp} \quad (6)$$

$$p_o = \left(1 - \frac{k_h}{k_d + k_h}\right) (TP - p_i) \quad (7)$$

$$p_p = \frac{k_h}{k_d + k_h} (TP - p_i) \quad (8)$$

Thus, we see that the ultimate inorganic phosphorus concentration is equal to the half saturation constant multiplied by the ratio of the phytoplankton loss rates ($k_r + k_d$) to the maximum net phytoplankton accrual rate ($k_g - k_r - k_d$). This implies that p_i will be on the order of k_{sp} ; that is, at a relatively low concentration. For lakes that are not light limited, this makes sense as phytoplankton should grow until they are nutrient limited.

Consequently, most of the initial mass of TP eventually resides in the organic and phytoplankton phases. The levels of these phases are then dictated by the relative values of hydrolysis and death as quantified by the dimensionless ratio, $k_h/(k_h + k_d)$. Thus, if recycle is efficient (hydrolysis is much faster than death), higher phytoplankton levels result. Conversely, if hydrolysis is relatively slow, the organic phosphorus builds up at the expense of the phytoplankton.

Although this is certainly a very simple model, it nicely illustrates why Vollenweider's approach might be considered sensible for the types of lakes he was investigating. In particular, because p_i should be negligible, Eq. (8) indicates a direct relationship between summer chlorophyll *a* (p_p) and spring TP.

Simple River Models

Whereas the foregoing model illustrates why specifying a total phosphorus concentration standard for lakes might be viable, it also serves to illustrate why it is ill-advised for rivers. This can be seen by recognizing that precisely the same model can be applied to rivers.

The river is idealized as a one-dimensional, plug-flow system with a single point source of phosphorus (Figure 5). Further it is assumed that the river has uniform, steady flow and constant hydrogeometric properties (i.e., depth, width, slope, etc.). For such cases, velocity will be constant and travel time and distance are linearly related (Thomann and Mueller 1987, Chapra 1997). That is,

$$t^* = \frac{x}{U} \quad (9)$$

where t^* = travel time (d), x = distance (m) and U = mean velocity (m/d).

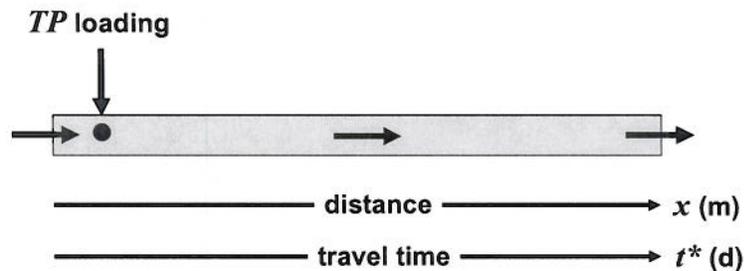


Figure 5 A one-dimensional, plug-flow representation of a river with a single point source of phosphorus.

For such a system, steady-state mass balances can be written for the three phosphorus species in Figure 3. By invoking the relationship of distance and travel time, the resulting mass balances are mathematically identical to the foregoing lake model (Eqs. 2 through 4). Just as for the lake, given reasonable values for the parameters and a set of initial conditions, these equations can be integrated numerically to simulate how the various phosphorus species change. But now rather than evolving in real time, the solution reflects how the species change spatially via travel time as we move downstream.

Using the same parameters and initial conditions as for the lake example (Table 1), the solution is as depicted in Figure 6. Notice that the patterns are identical to the lake (Figure 4) except that in the river the solution evolves in space downstream from the point source discharge.

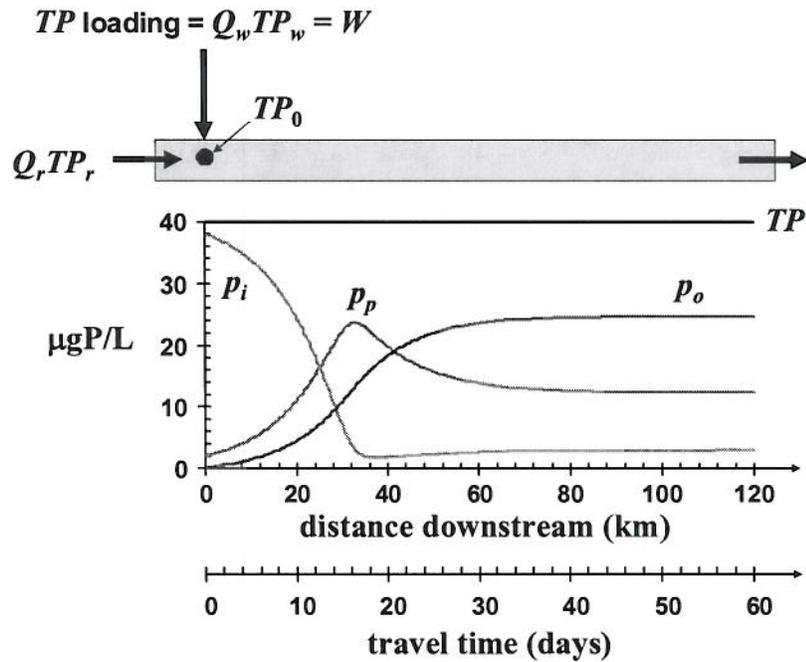


Figure 6 The spatial evolution of phytoplankton and phosphorus species in a river below a point source. The velocity of the river for this example is equal to 2 km/d as indicated by the travel time.

Although the equations, as well as the solutions, are identical, their interpretation is different. For example, the lake's initial conditions correspond to the spring phosphorus levels at the start of the growing season in time whereas the river's initial conditions represent phosphorus boundary conditions at the mixing zone in space. The river boundary condition for total phosphorus is computed as a flow-weighted average

$$TP_0 = \frac{Q_r TP_r + Q_w TP_w}{Q_r + Q_w} \quad (10)$$

where Q = flow rate (m^3/s), and the subscripts designate the mixing zone (0), the river upstream of the point source (r), and the point source (w). Note that for the purposes of TMDL determinations, the point source contribution is commonly expressed as a loading rate, $W = Q_w TP_w$. For cases where the point-source mass loading is much higher than the upstream river, Eq. (10) is often approximated as

$$TP_0 = \frac{W}{Q} \quad (11)$$

where Q = the total flow below the point source (m^3/s) = $Q_r + Q_w$. Thus, the initial condition represents the ratio of a mass loading rate to the river flow rate. The solution then describes how this initial mass loading rate is distributed among the three forms of phosphorus as the river flows downstream.

Such differences in interpretation suggest why setting a total phosphorus criterion might be sensible for a lake, but not for a river. For a lake, the major management focus is typically on the mid-summer period when recreational use is most intense. Hence, the ability to predict the summer chlorophyll level based on the spring TP concentration has management value.

In contrast for rivers, the peak phytoplankton level represents the principal impact. The solution in Figure 6 illustrates that until the stable asymptote is approached, there is no direct correlation between phytoplankton biomass and the total phosphorus concentration (as well as with any of the individual phosphorus species). Thus, just as is the case for BOD and oxygen, although phosphorus certainly causes increased phytoplankton biomass, there is no direct spatial correlation between in-stream TP and biomass in the vicinity of the worst eutrophication impact (i.e., the peak phytoplankton). Hence, while a phosphorus standard makes some sense for a long residence-time, stratified lake, it is less tenable for a plug-flow system like a river.

Whereas the foregoing mechanistic model illustrates the shortcoming of a total phosphorus concentration standard for river phytoplankton, it also suggests how the problem might be effectively managed. That is, the model can be used to calculate the concentration at the mixing point (i.e., the loading) so that the peak does not exceed a prescribed phytoplankton concentration standard.⁴ This can readily be accomplished numerically or for certain simplified situations, analytically.

⁴ This is directly analogous to managing the peak DO deficit for the oxygen problem.

References

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