# Assessment of the Scientific Basis of the Taunton Wastewater Treatment Plant Draft NPDES Permit (MA0100897)

by

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I have developed the following report as an assessment of the technical analyses used as the basis for establishing total nitrogen reduction requirements for the Taunton Wastewater Treatment Plant Draft NPDES Permit. In particular, my focus is on the scientific basis of (1) the total nitrogen concentration a (TN) criterion which was established as a target for bringing the system to acceptable water quality, and (2) the modeling methodology employed to generate the TN effluent limitations for discharges to the system's watershed. My review is based on careful reading of the permit and supporting documentation as well as a number of other relevant documents cited in the reference list at the end of this document. I begin with a summary of my understanding of the approach outlined in the permit. This is followed by a critique of specific aspects of the methodology. The report concludes with my overall assessment. My general conclusion is that the methods employed for developing the TN reduction requirements are not scientifically defensible and not consistent with the generally accepted methods used for assessing DO-related issues in estuaries. None of the important site-specific physical, chemical and biological factors influencing whether and how TN may affect the DO regime by stimulating excessive plant growth in the Taunton estuary were evaluated in the Region's analyses. Because overly simplistic, unreliable methods were employed in developing the permit requirements, there is no reasonable basis to conclude that the TN reduction requirements are either necessary or sufficient to ensure DO criteria compliance in the estuary. Consequently, absent a more complete and competent analysis that accounts for well-known factors influencing the DO regime in estuarine settings, the ecological benefits associated with TN reduction cannot be determined for this system.

## *OVERVIEW OF APPROACH*

Three types of empirical analyses are conventionally employed to derive numeric criteria for natural receiving waters (primarily lakes, rivers and estuaries): (1) the reference condition approach, (2) stressor-response analysis, and (3) mechanistic modeling (US EPA 2000a, 2000b, 2001, 2010b). In brief:

- The *reference condition approach* derives candidate criteria from observations collected in reference waterbodies representing least disturbed and/or minimally disturbed conditions within a region (Stoddard et al. 2006) that support designated uses.

- A *stressor-response analysis* is used when data are available to accurately estimate a relationship between nutrient concentrations and a response measure that is directly or indirectly related to a designated use of the waterbody. Then, a nutrient concentration that is protective of designated uses can be derived from the estimated relationship.
- *Mechanistic modeling* is used to predict specific constituents based on a series of equations and algorithms that represent physical, chemical, biological, and ecological processes. Thus, in contrast to the other two methodologies, which are empirical, the mechanistic models are based on scientific principles.

The Taunton TN nutrient criterion is based on a hybrid of the reference condition and stressor-response analysis, whereas a mechanistic model of sorts (albeit a very simple one) is used for the effluent limit calculation. For the former, a single “sentinel” station was chosen in Mount Hope Bay where DO criteria were met, and assumed that whatever TN level occurs at that location is what is the factor controlling the DO regime and therefore required to meet DO objectives throughout the system, including the Taunton estuary (many miles away). Thus, as with the reference condition approach, the current methodology uses data from a location that is deemed to have acceptable water quality and the physical factors influencing the DO regime are considered identical in all other locations. As with the stressor-response approach, the method is based on the implicit assumption that response (DO concentration) is well correlated with the stressor (TN concentration) and that no other significant factors are controlling the resultant water column DO.

Once the TN nutrient criterion was established, an estimate for the allowable TN loading was computed with a mass-balance. A very simple model was employed for this purpose. It was assumed that the entire estuary system was well-mixed and at a steady state. Given estimates of freshwater inflow rate and salinity concentrations, a salinity balance was then used to estimate exchange with the ocean. Given the ocean exchange, the model could then be used to compute the TN concentration of the freshwater inflow needed to achieve the TN concentration target. The product of the inflow rate times the inflow TN concentration then yields the allowable TN loading. Aside from its simplicity, the most noteworthy feature of the model is that it treats total nitrogen as a conservative substance.

## *CRITIQUE*

The methodology has many critical flaws which render its results thoroughly unreliable. These are the same type of fundamental flaws which were identified in the development of TN reduction requirements for the Great Bay estuary (Bierman, et al – 2014)) and which were identified by EPA’s Science Advisory Board in 2010 (US EPA 2010a) in reviewing the use of simplified regression methods to predict water quality and ecological changes due to ambient nutrient levels. Many of these deficiencies have already been identified by Hall and Associates (2014) with which I am in general agreement. Consequently, rather than reiterate the same points, my critique will focus on the flaws I found to be the most serious and fundamentally significant.

## Inappropriateness of Sentinel Method

There are a number of reasons why the sentinel method employed to come up with the nutrient criteria is fundamentally flawed and ultimately I have no expectation that meeting the ambient criteria chosen via this method will result in acceptable water quality throughout the system. First, it needs to be understood that this approach created to derive the Taunton permit requirements is novel and not specified as a scientifically defensible method for addressing DO-related problems in any published literature that I am familiar with in my 42 years of conducting water quality impact assessments. TN is not a pollutant that directly controls water column DO in estuarine systems. Therefore, as an initial point, the claim that simply controlling to achieve a specific TN level will produce a specific DO response is simply a false and scientifically incorrect assumption.

Second, both the reference condition and the stressor-response approaches are typically based on data from a number of similar systems. Statistical techniques are then employed to determine the most likely value of the nutrient criteria that correlates with acceptable water quality, after making sure that the system locations and physical factors are similar. The use of multiple systems and screening to ensure similar habitat and physical conditions (hydrodynamics and hydrology), greatly increases the reliability that the resulting nutrient criteria is generally valid and not the result of an outlier. In contrast, the use of a single station by the present study without any documentation that the other locations of the estuary are similar in hydrology/hydrodynamics and other critical factors (e.g., stratification and sources of DO demand) provides little confidence that the oxygen objective will be met at all (or even any) locations in the system. This is precisely the type of simplified analyses that EPA's Science Advisory Board informed the Agency was not sufficient or scientifically defensible in developing nutrient criteria and nutrient management approaches:

> "*For criteria that meet EPA's stated goal of "protecting against environmental degradation by nutrients," the underlying causal models must be correct. Habitat condition is a crucial consideration in this regard (e.g., light [for example, canopy cover], hydrology, grazer abundance, velocity, sediment type) that is not adequately addressed in the Guidance. Thus, a major uncertainty inherent in the Guidance is accounting for factors that influence biological responses to nutrient inputs. Addressing this uncertainty requires adequately accounting for these factors in different types of water bodies. (SAB report at 38) … Numeric nutrient criteria developed and implemented without consideration of system specific conditions (e.g., from a classification based on site types) can lead to management actions that may have negative social and economic and unintended environmental consequences without additional environmental protection.*" *(SAB at 38)* (US EPA 2010a)

The sentinel approach is predicated on the assumption that the total nutrient concentration at a single location provides a valid predictor of the dissolved oxygen concentration directly below that location and is similarly controlling the DO regime in other locations. Even for standing waters, like lakes, where vertical transport usually dominates, this is a tenuous assumption. For a flowing system such as an estuary, it is ludicrous. As is well documented in the literature, the oxygen at any estuarine location depends on a variety of factors including oxygen reaeration, depth, sediment oxygen demand, sediment-water exchange of nutrients, nitrification and denitrification, point source carbonaceous and nitrogenous loadings, degree of vertical mixing, horizontal transport from both upstream and downstream directions, algal productivity, hydrolysis, organic carbon and organic nitrogen loads from allochthonous sources in the watershed, etc., etc., etc. The failure to evaluate and consider any of these factors renders the present assessment pure speculation, which is, in an event, demonstrably in error. TN could not possibly be the single factor controlling the DO regime in the Taunton estuary given the numerous non-nutrient factors known to influence this and other estuarine systems.

## Choice of TN as stressor

The use of total nutrients as a stressor dates back to the early years of eutrophication modeling when Richard Vollenweider hypothesized that the spring total phosphorus concentration in a lake could be used as a predictor of summer eutrophication symptoms such as average chlorophyll *a*, Secchi depth, and hypolimnetic dissolved oxygen demand (Vollenweider 1968, 1969, 1975). This made some sense for stratified lakes with low to moderate summer flushing rates as the lake's surface layer could be viewed as a batch reactor. However, Vollenweider and other water-quality experts recognized that although the approach could be used for crude screening analysis of stratified lakes, more sophisticated methodologies would be required for actual management of other water bodies such as shallow lakes, and flowing systems such as rivers and estuaries.

Because they are subject to strong advective water motion, flowing systems (such as rivers and estuaries) are the antithesis of batch systems and hence, the idea that a total nutrient will ultimately and predictably yield a particular level of water quality at a point in space and time is again patently ludicrous. I have included an appendix at the end of this document, where I use simple mathematical models to illustrate why this is true.

## Oversimplistic Modeling

As mentioned previously, no water quality modeling was employed to establish the reliability of the TN criterion. At a minimum, the analysis should have demonstrated how TN influenced phytoplankton growth at the various locations, since this is a prerequisite for causing effects on the DO regime. No such analysis exists. Because of the complexity of this system and its economic and environmental value the absence of any serious modeling to support nutrient criteria development verges on negligent.

Further, even when modeling is employed to establish the TN effluent limitations, it is ludicrously simplistic and based on completely undocumented assumptions, rather than scientific fact or an exercise of reasonable scientific judgment. First, the system is clearly not completely mixed with gradients occurring longitudinally, laterally and vertically. Second, eutrophication is not a steady-state problem as is clearly demonstrated by the time series plots contained in the permit document. The very fact that phytoplankton "blooms" occur establishes that the systems water quality is dynamic. Third, the assumption that TN is conservative is absolutely erroneous. Although it is clearly more stable than its component species (e.g., ammonia, nitrate, etc.), a number of source and sink processes act to increase and reduce the total nitrogen pool at different rates in different locations in the system. Notable among these are sediment-water interactions (settling, resuspension, sediment nutrient release) and denitrification. Finally, there is no rational basis to presume that the important hydrodynamic conditions controlling the DO regime and how TN may influence that regime are identical in Mount Hope Bay and the upper reaches of the Taunton estuary. This is pure speculation which is, once again, demonstrably incorrect as the hydrodynamic and hydrologic conditions in these two areas are obviously quite different as would be expected by simply looking at a map of the estuary and given a rudimentary understanding of coastal hydrodynamics (one is the closed end of the Taunton estuary affected by fresh water inputs, the other would be primarily influenced by higher tidal exchange from the ocean). In short, the "modeling" has no credible scientific basis.

### ***SUMMARY***

In summary, I have concluded that the technical analysis underlying the permit is severely flawed, and does not reflect the current or accepted state of the science for making such assessments. It is based on naïve and simplistic reasoning that is weak and clearly not consistent with the available information or expected conditions controlling the DO regime in estuarine settings. No published EPA guidance document on assessment of DO and nutrient conditions in estuarine settings indicates that this is an accepted method of analysis.

I have critiqued many water quality plans and management schemes as an environmental engineer and water-quality expert and I must state that this is the most technically weak effort I have examined over my 42 year career. (See attached curriculum vitae). And lest my comments be considered biased, I should state that beyond my scientific background, I am a dedicated environmentalist who was drawn to this field because of my love of the outdoors. I have fished the New England coastline from Long Island Sound off New London to north of Cape Ann in Massachusetts and I believe that Narragansett Bay is one of the real jewels of our region. So it really matters to me that the stewardship of systems such as the Taunton River Estuary and Mount Hope Bay be based on the best available science. Because this is not the case, I have absolutely no confidence that the remedial measures suggested by the permit will have the desired effect of maintaining healthy water quality in the system.

Steven C. Chapra **Sept. 5, 2014**

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## *APPENDIX 1. Why TP Concentration Standards are Inappropriate for Managing Phytoplankton Biomass in Flowing Systems*

This appendix attempts to address the question of why anyone would ever suggest that a total phosphorus criterion would represent a sensible strategy for managing flowing systems such as rivers or estuaries. In brief, I believe that the idea of singular total phosphorus criteria for flowing natural waterbodies originates from the misguided notion that effective lake management approaches can be seamlessly (and thoughtlessly) transferred to rivers and streams. Although the following focuses on phosphorus in rivers, the conclusions are directly transferable to nitrogen-limited tidal rivers and estuaries.

In the late 1960’s and early 1970’s, several limnologists suggested that total phosphorus concentration could serve as an effective trophic state indicator. 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 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.

In effect, the viability of the Vollenweider approach is predicated on the functioning of the particular lakes he studied. In particular, the approach was developed for deep, stratified, phosphorus-limited, North-temperate lakes 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.

A simple computation can be used to illustrate how such an approach breaks down in rivers and streams. First, total phosphorus can be divided into three components

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

where $p_p$ = phytoplankton phosphorus (μgP/L), $p_i$ = inorganic phosphorus (μgP/L), and $p_o$ = non-phytoplankton organic phosphorus (μgP/L). If the chlorophyll *a* to phosphorus ratio is assumed to be 1 μgA/μgP, this means that $p_p$ can be directly interpreted as a measure of phytoplankton biomass.

The river can be idealized as a steady-state, plug-flow system with a single point source of phosphorus (Figure 1). Further it is assumed that the river has uniform, steady flow and constant hydrogeometric properties (i.e., depth, width, etc.). For such cases, velocity will be constant and travel time and distance are linearly related (i.e., distance = velocity times travel time). Under these conditions, the following mass-balances can be written for each phosphorus component

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

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

$$\frac{dp_o}{dt} = k_d p_p - k_h p_o \tag{4}$$

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

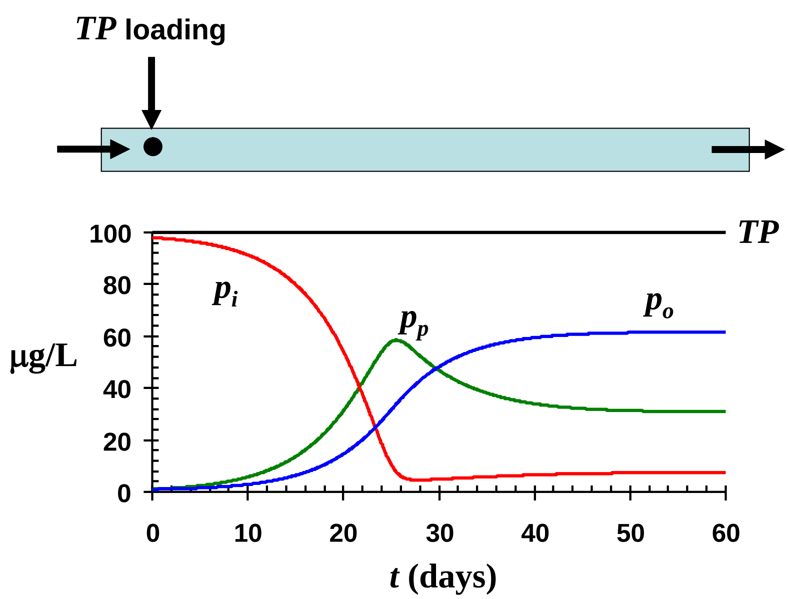


**Figure 1 Simulation of phytoplankton, inorganic and organic phosphorus downstream from a point source.**

Given reasonable values for the parameters and a set of initial conditions at the mixing point (Table 1), these equations can be integrated numerically to simulate how the various phosphorus species change as the water travels downstream. For the present example, the initial conditions are set so that the river has a high level of available, inorganic nutrient at the mixing point as would be the case for a high phosphorus discharge into an effluent-dominated river. In addition, the phytoplankton settling velocity is set to zero.

**Table 1 Parameters and initial conditions used to simulate phytoplankton and phosphorus concentrations below a single point source to a one-dimensional river.**

| Parameter | Value | Units |
|---|---|---|
| $k_g$ | 0.5 | $d^{-1}$ |
| $k_{sp}$ | 5 | $\mu gP\ L^{-1}$ |
| $k_r$ | 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 | $\mu gP\ L^{-1}$ |
| $p_i$ | 98 | $\mu gP\ L^{-1}$ |
| $p_o$ | 1 | $\mu gP\ L^{-1}$ |

The results are displayed in Figure 1. Because the inorganic P concentration is well above the half-saturation constant, the phytoplankton initially grow rapidly as the 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 the respiration and death losses. Thereafter, the phytoplankton levels decline until the solution approaches a stable steady state. This asymptote represents the point at which phytoplankton growth exactly balances phosphorus recycle.

Note that 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

$$p_i = \frac{k_r + k_d}{k_g - (k_r + k_d)} k_{sp} \tag{5}$$

$$p_o = \left(1 - \frac{k_h}{k_d + k_h}\right)(TP - p_i) \tag{6}$$

$$p_p = \frac{k_h}{k_d + k_h}(TP - p_i) \tag{7}$$

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 growth rate ($k_g - k_r - k_d$). The organic P and phytoplankton

P concentrations are then dictated by the product of the total organic P (i.e., organic P and phytoplankton P) and a dimensionless number quantifying the relative values of the hydrolysis and death rates.

Although this is a very simple model, it dramatically illustrates why specifying a phosphorus concentration standard for rivers is ill-founded. Notice that until the 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).

Just as is the case for BOD and oxygen, although phosphorus certainly causes increased phytoplankton biomass, there is absolutely no direct spatial correlation between in-stream TP and biomass. Hence, whereas a phosphorus standard makes some sense for a long residence-time, stratified lake, it falls apart for a plug-flow system like a river (or a mixed-flow system such as an estuary).