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National Association of Clean Water Agencies, Washington D.C.

Review of USEPA Methods for Setting Water Quality-Based Effluent Limits for Nutrients

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Dr. Denny Parker is a Senior Vice President and Director of Technology for Brown and Caldwell. Throughout his career he has served as a process designer or a reviewer for major nutrient removal plants. He has extensive experience in the introduction of innovative treatment technologies. Dr. Parker has invented, developed and implemented new wastewater processes that are now used in the wastewater industry in nutrient removal plants including a number of separate stage nitrification and denitrification processes and chemically enhanced primary treatment for phosphorus removal. His contributions to nutrient removal in the activated sludge process include the Classifying Selector eliminates nuisance foams from the process and the BAR process provides sidestream treatment by nitrification in a reaeration basin, charging the mainstream process with nitrifiers thereby reducing overall aeration tank requirements. He also developed the Biological Contact process which borrows mixed liquor from the mainstream process to provide high rate BOD removal for wet weather flows, to protect the mainstream activated sludge plant to ensure nutrient removal is sustained at all times. Dr. Parker has won eight national awards for his process engineering work including induction into the National Academy of Engineering in 2004, WEF's Camp Medal in 2003 and has been elected a WEF Fellow.

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Matthew DeBoer is an environmental engineer with 12 years of experience in assessing water quality via outfall modeling and design, conducting field studies, National Pollutant Discharge Elimination System (NPDES) permitting, and municipal and industrial wastewater facility planning. Matt is also experienced in the development of NPDES permit applications, facility plans/engineering reports, and mixing zone studies.

List of Abbreviations

AML	average monthly limit
ASIWPCA	Association of State and Interstate Water Pollution Control Administrators (now the Association of Clean Water Administrators)
ATP	adenosine triphosphate
BOD	biological oxygen demand
CCC	criteria continuous concentration
CMC	criteria maximum concentration
CV	coefficient of variance
DMR	discharge monitoring report
HAB	harmful algal bloom
LOT	Limit of Technology
LTA	long-term average
MDL	maximum daily limit
MDL	minimum detection limit
NACWA	National Association of Clean Water Agencies
NNC	numeric nutrient criteria
NPDES	National Pollutant Discharge Elimination System
POTW	publicly owned treatment works
RPA	reasonable potential analysis
TMDL	total maximum daily load
TN	total nitrogen
TP	total phosphorus
TSD	Technical Support Document for Water Quality- Based Toxics Control
TSS	total suspended solids
USEPA	United States Environmental Protection Agency
WERF	Water Environment Research Foundation
WET	whole effluent toxicity
WLA	wasteload allocation
WQBEL	water quality-based effluent limits

Executive Summary

Nutrient control is one of the chief regulatory and economic issues facing wastewater utilities today. USEPA's national strategy on nutrients has focused on the derivation of numeric nutrient criteria (NNC), and the use of those criteria to develop water quality-based effluent limits (WQBELs) in a manner analogous to what has been historically performed for toxic substances. For example, USEPA has not developed separate permitting guidance for nutrients, but has largely relied on the calculation methods of the *Technical Support Document for Water Quality-Based Toxics Control* (TSD). There has been a great deal of variability in the degree to which regulatory agencies have modified these methods to account for the differences between nutrients and toxics, and in many settings, no modification has been performed.

This report presents a technical review of federal guidance for deriving WQBELs for nutrients in National Pollutant Discharge Elimination System (NPDES) permits. The report was commissioned by the National Association of Clean Water Agencies (NACWA) as an independent review of which elements of federal toxics-based permitting methods are valid or invalid for nutrients, or would require modifications to be valid for nutrients. This review demonstrates that due to fundamental differences in how nutrients and toxics affect receiving waters, the use of toxics-based methods is often inappropriate. The review also seeks to identify potential alternatives where warranted. NACWA intends this review to facilitate ongoing discussions with agencies and stakeholders on regulatory solutions for nutrients.

Methods

The review of TSD-based permitting methods was conducted in two steps. In the first step, individual permitting elements were evaluated for appropriateness for nutrient permitting based on conceptual models of nutrient impacts, the scientific literature, and regulatory precedents. Based on the outcome of the review, each element of the toxics-based permitting approach was placed into one of the following categories:

- Usually appropriate for use with nutrients without modification.
- Appropriate for use with nutrients in certain circumstances, or with appropriate modifications.
- Usually not appropriate for use with nutrients.

If a toxics-based permitting element was deemed to require modification for nutrients, recommendations for such modifications were provided. Similarly, if a permitting element was deemed not appropriate for nutrients, recommendations were provided on alternative approaches. The review also identifies permitting elements that need to be included for nutrients, but are not fully addressed by the TSD or otherwise routinely considered for toxics permitting.

In the second step of the review, results of the evaluation were used to develop recommendations on a potential permitting framework for nutrients. These recommendations address the major considerations for nutrients regarding reasonable potential analysis, derivation of wasteload allocations (WLAs), WQBEL development, and the consideration of technical feasibility.

Results

The results of this review (summarized in Table ES-1) demonstrate that the procedures for developing WLAs and WQBELs for nutrients should differ from toxics permitting in profound ways. These differences are rooted in fundamental differences in the mechanisms of nutrient and toxic-related impairments, and the high degree of variability in how water bodies respond to nutrients. Unlike the dose-response effects expected from toxics, nutrient effects are often better characterized as indirect and water body-specific. Due to these differences, there are few elements of the toxics-based permit methods that do not merit some level of modification for nutrients.

Table ES-1. Summary of TSD Component Review Results

Toxics-Based Permitting Component	Result of Evaluation	Explanation
Assumption of dose-response; acute and chronic concepts	○	Nutrient effects are indirect, gradational, water body-specific.
Human health concepts	○	Nutrients do not bioaccumulate, are not carcinogenic, and are not directly toxic except for certain species at relatively high levels.
Focus on concentration targets	○ / ⦿	Valid concentration targets may be available in some settings; for others, load is more meaningful.
Excursion frequency for protection of aquatic life	⦿	1-in-3 year frequency may be appropriate if longer averaging periods are used.
Critical conditions	⦿	Should be more common and less transient that 1Q10/7Q10 conditions used for toxics.
Reasonable potential analysis based on 95th to 99th percentile	○	Nutrient impacts not controlled by short-term spikes.
Deriving WLAs from steady state models	⦿	Appropriate in limited circumstances. Steady state condition must be modified to reflect appropriate critical condition.
Deriving WLAs from dynamic models that link designated uses to nutrient concentrations	●	Preferred for complex water bodies.
Consider downstream effects	●	Nutrient WQBELs must protect downstream uses.
Consider variability in effluent characteristics when setting WQBELs	⦿	Effluent variability should be considered implicitly or explicitly.
Assumption of log normality	⦿	Verify with facility-specific data.
Method for calculation of the long-term average	⦿	Modify to reflect longer averaging periods of WLA. Probability basis can be lower than with toxics.
Expressing limits as maximum daily limits	○	Daily time frames not meaningful for nutrients.
Expressing limits as longer averaging periods	⦿	Monthly, seasonal, or annual limits are appropriate, depending on the receiving water body dynamics.
Relevance of # samples per averaging period	⦿	Using longer averaging periods, the sensitivity of limits to sample number is less. Can be considered implicitly.
Mixing zone concepts	⦿	Use a full mix when protecting against far field effects. Near field effects could merit alternative mixing/assimilation zone concepts.

Key: ○ Rarely appropriate for nutrient permitting.

⦿ Appropriate for nutrient permitting in certain circumstances, with recommended modifications to the TSD method.

● Usually appropriate for nutrient permitting.

For example, TSD's basic approach for conducting a quantitative reasonable potential analysis for toxics is not applicable to nutrients. Differences between nutrients and toxics also point toward factors such as longer averaging periods, different critical conditions, different application of mixing concepts, and a preference for load-response predictions over the use of default concentration targets. Unlike most toxic parameters, nutrients are contributed by almost every point and nonpoint source on the landscape, requiring a closer consideration of equity, cost-effectiveness, and treatability.

It is the authors' view that, after a WLA is derived that links designated uses to nutrients, the TSD's basic statistical framework for calculating WQBELs from WLAs could be successfully modified for nutrients in many circumstances. Key modifications would include verification/modification of the assumption of lognormality, application of different (i.e., longer) averaging periods, and selection of probability bases that reflect a level of protection that is more appropriate to nutrients. The direct use of WLAs as WQBELs and actual probability distributions for short term effluent variability represent viable alternatives to the TSD method for addressing effluent variability (e.g., 95 and 99% allowable for short periods).

Recommendations on a Nutrient Permitting Framework

WQBEL calculation is one arena where the differences between nutrients and toxics merit alternative approaches. However, it is also helpful to step back from calculation details and consider the wider implications for a nutrient permitting framework. In broad terms, it is recommended that nutrient permitting procedures include more WLA/limit development flexibility, specifically to address the greater nutrient effluent variability and of the appropriate timeframes associated with nutrients effects in the environment. The authors recognize that permitting is the responsibility of both USEPA and states, and nutrient permitting approaches would vary based on hydrologic settings and state-specific regulations. However, it is possible to identify broad recommendations that would aid the development of sound WLAs and limits in most settings (Table ES-2). Many of these recommendations require departures from the toxics paradigm to one degree or other, but all can be accomplished within the Clean Water Act framework.

This review document concludes by describing permitting framework for nutrients that incorporates many of the recommendations above. Some aspects are similar to the Florida Department of Environmental Protection's (DEP) (2013) proposed approach for implementing nutrient standards, which provides several alternatives (other than default concentration targets) for developing nutrient permits, including:

- The use of existing TMDLs;
- Basing limits on existing conditions for biologically healthy water bodies;
- Developing load-response linkages.

The conceptual permitting framework includes nutrient-specific considerations and the option for using non-WQBEL approaches if standard methods would result in unattainable limits with uncertain benefits. The framework also includes three options for calculating WQBELs from WLAs. Options discussed in this report include the use of a modified TSD approach, setting WQBELs to WLAs, or using empirical distribution functions. Properly applied/modified for nutrients, all three of these methods are capable of giving similar WQBELs. Differences between nutrients and toxics modes and timeframes of action must be considered at every permitting step, from WLA to reasonable potential analysis to WQBEL calculation. It is hoped that this review and recommendations will provide a technical basis for a useful discussion of these differences, and help advocate viable nutrient permitting framework.

Table ES-2. Summary of Recommendations	
Category	**Recommendation**
Deriving Wasteload Allocations	1. Recognize load-response linkages as a viable alternative to the use of concentration targets for deriving WLAs, depending on the water body.
	2. Provide options for use of biological confirmation to inform WLAs, such as not reducing WLAs for healthy receiving waters.
	3. Consider preferential nutrient controls (i.e., greater control of limiting nutrient) to attain desired responses and reduce compliance costs.
	4. Use critical conditions and frequency components that are tailored to nutrient responses. These will generally be less rare and transient than critical conditions commonly used for toxics (e.g. 7Q10 streamflow)
	5. Assume full mix for nutrients when they are being controlled to prevent far-field impairments. For near-field effects, explore alternative mixing/assimilation zone concepts.
	6. Use watershed-based permitting approaches that consider equity, cost-effectiveness, and ancillary effects directly in the WLA derivation process, and support the use of trading/offsets.
Calculating Water Quality Based Effluent Limits	7. Use the appropriate averaging periods for deriving and expressing limits. Seasonal or annual averaging periods can be appropriate, particularly for receiving water bodies with longer retention times, or when using criteria with longer duration components.
	8. Verify or modify assumptions regarding effluent variability (e.g., lognormality and coefficient of variation) based on facility or specific control technology-specific data.
	9. Choose probability bases that reflect a reasonable level of conservativeness for nutrients, based on the mode and timing of nutrient effects.
	10. Consider using WLAs as WQBELs. When using longer averaging periods, this approach is conservative, and can better support watershed-based control strategies including trading programs.
	11. Consider empirical alternatives to the TSD approach for calculating WQBELs, particularly if monitoring data are available and depart from log normal assumptions.
Considering Treatability Limits	12. Do not set WQBELs to levels that cannot be reliably attained.

Section 1

Introduction

This report presents a technical review of federal guidance for deriving water quality-based effluent limits (WQBELs) for nutrients in National Pollutant Discharge Elimination System (NPDES) permits. The report was commissioned by the National Association of Clean Water Agencies (NACWA) as an independent review of the validity of federal toxics-based permitting methods as applied to nutrients. The review also seeks to identify potential alternatives. For the purposes of this report, “valid” water quality-based permitting approaches are defined as those that quantitatively link targets/limits to designated use attainment with a reasonable degree of uncertainty and conservativeness. NACWA intends this review to facilitate ongoing discussions with agencies and stakeholders on development and implementation of a regulatory approach that is appropriate for nutrients.

1.1 Background

Since passage of the Clean Water Act, the United States Environmental Protection Agency (USEPA) and states have developed various guidance documents for the development of limits for NPDES permits. Of these, none has been more important than the Technical Support Document for Water Quality-Based Toxics Control (USEPA, 1991), abbreviated herein simply as the TSD. This document describes statistical procedures for determining when permits limits for toxics are necessary, and for calculating both mass-based and concentration-based WQBELs from wasteload allocations (WLAs) if such limits are deemed necessary. Most states reference this document heavily in the development of their own NPDES permitting procedures.

1.1.1 Use of Toxics-Based Permitting Approaches for Nutrients

The TSD was written specifically for developing permit limits for toxic substances such as metals, organic compounds, and whole effluent toxicity (WET). The manner in which nutrients affect receiving water bodies is different from toxics in fundamental ways. However, USEPA has not issued equivalent guidance for determination and development of WQBELs to protect waters from nutrient-caused impacts. Rather, USEPA and some states have defaulted to using the TSD method for nutrient WQBELs. In some permitting situations, agencies have modified the methods in an attempt to account for differences between nutrients and toxic impacts. In other permitting situations, little to no modification has occurred.

In July of 2013 USEPA conducted a Permit Writers Specialty Workshop for state regulatory agencies that administer the Clean Water Act. The content of the workshop included a training presentation entitled “Developing WQBELs (Water Quality Based Effluent Limits) for Nutrient Pollution.” This workshop introduced to state permit writers the concept of using the TSD for the translation of state narrative standards to numeric standards for nutrients. The training slides include content on ways by which some aspects of the TSD approach could be adjusted for nutrients. However, USEPA has not presented a comprehensive permitting framework in either the training slides or other document for nutrients that is fundamentally different from toxics. Instead, much of USEPA’s National Nutrient Strategy has focused on the derivation of numeric nutrient criteria (NNC), and the use of those criteria to develop WQBELs in a manner similar to that performed for toxics.

The development of numeric nutrient criteria has proven to be challenging in many of the nation's waters. Understanding the significant complexities of when and how nutrients affect a water body have prevented the development of numeric criteria in most states. This has caused many states to continue to rely upon narrative or qualitative criteria, or to seek alternatives to the toxics-based paradigm for criteria development, assessment, and permitting.

1.1.2 Applicability of 40 CFR 122.44(d)

In 1987, USEPA amended federal regulations by adding section 122.44(d) which addresses how states will create numeric water quality standards for toxic pollutants. Subsection (1)(vi) specifically requires "…Where a State has not established a water quality criterion for a specific chemical pollutant that is present in an effluent at a concentration that causes, has the reasonable potential to cause, or contributes to an excursion above a narrative criterion within an applicable State water quality standard, the permitting authority must establish effluent limits using one or more of the following options…", which can be summarized as:

1. Using a numeric water quality criterion established by the permitting authority.
2. Using water quality criteria published by USEPA, on a case-by-case basis.
3. Using an indicator parameter for the pollutant of concern.

At the time of the rule change, toxic pollutants were of increasing concern, as an expansion of the prior focus on conventional pollutants such as total suspended solids (TSS) and biological oxygen demand (BOD). At the time, USEPA's primary focus was on toxics and WETrather than nutrients.

Since its inception, section 122.44(d) and implementing guidance (TSD) has been used by states and USEPA to include effluent limits for toxics pollutants in NPDES permits. Much of the success and acceptance of toxics regulations has been due to the demonstrable and reproducible association of the concentration of toxic substances with effects on living resources. However, the manner in which nutrients affect designated uses is much more complex and water body-specific than with toxics. In addition, nutrients are derived from a much wider spectrum of sources than toxics, and the Clean Water Act has not been used effectively to control nonpoint source nutrient contributions. USEPA's national strategy for addressing nutrients has persistently been to encourage states to derive NNC and use them for assessment and permitting. USEPA's major nutrient-related publications and communications since the 1990s have included:

- 1998: The National Strategy for the Development of Regional Nutrient Criteria, which stated USEPA's intent to derive regional criteria and water body-type guidance documents, and USEPA's expectation that states would adopt numeric nutrient standards.
- 2000-2001: Ecoregional criteria documents for lakes/reservoirs, rivers/streams, and wetlands. The criteria presented in these documents were derived using simple percentile-based approaches.
- 2001: A policy memo entitled "Development and Adoption of Nutrient Criteria into Water Quality Standards" by Geoffrey Grubbs. This memo emphasized USEPA's expectation that states would develop and execute plans to establish quantitative endpoints for nutrients. It stated that both reference conditions approaches and quantitative, predictive approaches (including the use of narrative criteria with translators) are valid approaches.
- 2000-2002: Technical guidance manuals for deriving nutrient criteria in rivers/streams, lakes/reservoirs, and estuaries. These manuals emphasize the use of reference conditions and simple percentile-based approaches for criteria development, but also touch on the potential roles of more predictive approaches.

- 2010: A technical memo entitled "Using Stressor-Response Relationships to Derive Numeric Nutrient Criteria", which describes various empirical approaches for exploring the statistical relations between nutrients and response variables.
- 2011: A policy memo entitled "Working in Partnership with States to Address Phosphorus and Nitrogen through Use of a Framework for State Nutrient Reductions" by Nancy K. Stoner. This memo listed eight recommended elements of a state framework to address nutrient pollution, including activities such as watershed prioritization, point source permitting, development of watershed-scale plans, and nutrient criteria development.
- 2013: A document entitled "Guiding Principles on an Optional Approach for Developing and Implementing a Numeric Nutrient Criterion that Integrates Causal and Response Parameters". This document endorses the concept of the joint use of causal (N and P) and response variables for assessment, with caveats.

As early as January of 2011, USEPA attempted to have states utilize 40CFR 122.44(d)(1)(vi) in the creation of numeric nutrient criteria. In a letter from USEPA Region 5 to the State of Illinois, the agency instructed that, in the absence of NNC, a reasonable potential determination should be made based on the state's narrative nutrient criteria. The letter stated that, where a discharger has the potential to cause or contribute to an exceedance of the narrative criteria, the state should develop a permit limit using one of the options in §122.44(d)(1)(vi). Following its letter to the State of Illinois, in March of 2011, USEPA issued the "Recommended Elements of a State's Nutrient Framework" in an attempt to encourage a cooperative regulatory approach between the agency, its regional offices, and the states.

More recently, NPDES permits issued in USEPA Region 1 have included numeric nutrient limits utilizing the approach from 40 CFR 122.44(d)(1)(vi). In these cases the permits were issued by the agency as the permit authority in non-delegated states. Permits in both New Hampshire and Massachusetts are presently being appealed at least in part due to the use of this methodology.

In a March 2013 letter to the agency, NACWA recommended that USEPA discontinue efforts to utilize 40CFR 122.44(d)(1)(vi) in the manner described above. NACWA suggested that the proper approach was for the agency to assist states in the implementation of numeric nutrient criteria that incorporated "meaningful dialog between stakeholders." Alternatively, if the Agency persists in the use of Section 122.44(d)(1)(vi), NACWA recommended that it be through a formal guidance development process that includes a dialog with permittees and other stakeholders.

1.2 Purpose and Scope

This document is intended to provide a technical review of the appropriateness of the TSD approach in the development of WQBELs for nutrients. Included herein are discussions on the basic differences between toxics and nutrient pollutants (Section 2) and the characteristics of nutrients in wastewater effluent (Section 3). The heart of the report is a detailed review of individual TSD-based permitting components in the context of nutrients (Section 4). The final sections of the report provide a summary of findings and high-level recommendations for development of a viable nutrient permitting framework (Section 5) and references (Section 6).

1.3 Materials Subject to Review

This review utilized a large number of references, including USEPA guidance, state permitting guidance, total maximum daily load (TMDL) reports, prior permitting precedents, and the scientific literature including Water Environment Research Foundation (WERF) reports. These references are cited in the report body, as needed, to support specific findings. However, there are three references (the TSD, Permit Writers' Manual, and 2013 training materials) that merit additional descriptions, because they represent the primary subject of the review.

1.3.1 USEPA Technical Support Document

In 1991, USEPA's Office of Water issued the Technical Support Document for Water Quality-Based Toxics Control, USEPA/505/2-90-001. As stated in the document's forward, the document was intended to provide technical guidance for assessing and regulating the discharge of toxic substances to the waters of the United States. It can be accessed on the internet at:

http://www.epa.gov/npdes/pubs/owm0264.pdf

This document was intended to provide the states and regions with guidance on how to appropriately control toxic substances in the nation's waters. This document is a revision to the original TSD issued in 1985. This revision became necessary when revisions to the CFR occurred in 1987 with an emphasis on controlling toxics.

1.3.2 USEPA Permit Writer's Manual

In 2010, USEPA issued the current edition of the NPDES Permit Writers Manual, providing a further update to the two previous versions (1993, 1996). The guide is intended to assist new permit writers in becoming acquainted with the NPDES program and to provide technical background for permitting decisions in delegated states and the regions. It can be accessed on the internet at:

http://www.epa.gov/npdes/pubs/pwm_2010.pdf

The manual has very little content that is specific to nutrient permitting and, therefore, may be most notable for not drawing many distinctions in how nutrients and toxics would be permitted. It does include a single paragraph that identifies some of the differences between USEPA's ecoregional nutrient levels and toxics criteria, and indicates that states may adopt seasonal or annual averaging periods for nutrients. The manual also specifically recommends that the non-conservative or nutrient cycling behavior of nutrients be considered during the permitting process.

1.3.3 USEPA Training Materials for Nutrient Permitting

In 2013, USEPA developed training materials entitled "Developing WQBELs (Water Quality Based Effluent Limits) for Nutrient Pollution." The materials were developed for an NPDES Permit Writer Conference for state regulatory agencies that administer the Clean Water Act, and are in the form of a slide presentation. The introduction states that the presentation "…supplements, and does not modify, existing USEPA policy, guidance, and training on NPDES permitting." Given the lack of nutrient-specific permitting guidance, the slides represent the most detailed recommendations available from USEPA on how nutrient WQBELs should be developed.

The presentation is structured around the typical NPDES permitting sequence of identifying applicable water quality standards, determining the need for WQBELs, calculating WQBELs, and determining final effluent limitations. The recommended calculation procedures are largely analogous to the TSD statistical approach, and are largely dependent on the availability of in-stream nutrient concentration targets. Where state numeric nutrient criteria are lacking, the materials describe the selection of default concentration targets, such as USEPA ecoregional criteria or 1986 Gold Book values. However, the USEPA training materials do directly acknowledge many of the conceptual differences between nutrients and toxics that would affect permitting. For example, slide 19 is titled "Why Not Use the TSD?" and states that nutrients present concerns that require considerations beyond the TSD procedures. Among the considerations listed are:

- Interpretation of nutrient-related narrative criteria.
- Duration and frequency of nutrient criteria.
- Critical effluent and receiving water conditions.

- Water quality modeling for nutrients.
- Determination of the need for WQBEL-based numeric and narrative nutrient criteria.
- Calculation of WQBELs with appropriate magnitudes and averaging periods for nutrients.

On various slides, the presentation provides examples of permit calculations. Some of these examples explicitly recommend modification of the TSD approach for nutrients, especially regarding the potential use of longer averaging periods for limit calculation or expression. Although the slides touch on concepts such as deriving WLAs from models and adjusting critical conditions for nutrients, it was beyond their scope to provide detailed guidance on how these permitting elements would differ between nutrients and toxics.

1.4 Review Approach

The review of TSD-based permitting methods was conducted in two steps: (1) review of individual permitting components; and (2) development of recommendations specific to a science- or technology-based nutrient permitting framework. These two steps are described in subsections below.

1.4.1 Review of Individual Permitting Components

First, the project team conceptually deconstructed the toxics-based permitting approach into individual components and concepts for review as applied to nutrients. These include components such as critical conditions, assumptions of dose-response, statistical distributions, duration/frequency of impacts, manner of considering variability, etc. These elements were evaluated for appropriateness for nutrient permitting based on conceptual models of nutrient impacts, the scientific literature, and regulatory precedents. Based on the outcome of the review, each element of the toxics-based permitting approach was placed into one of the following categories:

- Usually appropriate for use with nutrients without modification.
- Appropriate for use with nutrients in certain circumstances, or with appropriate modifications.
- Usually not appropriate for use with nutrients.

If a toxics-based permitted element was deemed to require modification for application to nutrients, recommendations for such modifications were provided. Nutrient permitting situations could cover a very large range of hydrologic and regulatory settings, and it was recognized that some elements of the toxics-based permitting methods might be more appropriate—or require less modification—in some settings than in others. Similarly, if a permitting element was deemed not appropriate for nutrients, recommendations were provided on alternative approaches. The review also identifies permitting elements that need to be included for nutrients, but are not fully addressed by the TSD or otherwise routinely considered for toxics permitting.

1.4.2 Recommendations on Nutrient Permitting Framework

After the review of individual permitting components, the second step of the review was to provide recommendations on developing an overarching permitting framework for nutrients. These recommendations address the major considerations for nutrients regarding reasonable potential analysis, WLA derivation, WQBEL development, and the consideration of technical feasibility. The summary includes a decision tree that frames some of the key questions that permitters would face in determining the best nutrient permitting approach in a given situation. These recommendations could potentially serve as a starting point for the development of nutrient-specific permitting guidance.

Section 2

Comparison of Nutrient Impacts with Toxic Impacts

Increased availability of nutrients such as nitrogen and phosphorus, in combination with other environmental factors, can lead to excessive algae and plant growth in streams and lakes. This can have negative effects on aquatic life, recreation and water supply uses of these waters. However, impacts from nutrients are markedly different than impacts from the more traditional toxic parameters around which the TSD was developed. The paradigm of toxics impacts is a dose-response evaluation for individual pollutants that may have adverse impacts on aquatic life or human health. Some individual nutrient species, such as ammonia, can have direct toxic impacts and have separate water quality criteria to prevent toxic effects. However, most nutrient management efforts are not focused on the direct effect of nutrients, but at indirect impacts that manifest themselves through a variety of response variables such as aesthetics (e.g., excessive algal biomass, or lack of water clarity) or aquatic life effects (e.g., low dissolved oxygen). These effects are often water body-specific and depend in part on non-nutrient factors such as light availability, carbon dioxide, temperature, pH, flow quantity and velocity, species of phytoplankton, salinity, sediment, other trace components, organic matter, and other factors (Bell, 1992; Yang, 2008). This section compares and contrasts toxic impacts and nutrient impacts with regard to mechanisms and timing.

2.1 Aquatic Life

Based on the TSD, the toxic impacts for aquatic life are a dose-response based determination of the concentration of a pollutant that will cause lethality or growth and reproductive effects. A specific concentration will evoke a direct response in an organism, and standards or permit limits are set to protect against that concentration. Toxic impacts may be seen fairly rapidly (hours to days), and the direct effects of exposure to a certain parameter is based on a given duration, magnitude, and frequency. Magnitude, frequency, and duration components of criteria are evaluated based on toxicity testing and other available information regarding the toxicity of a specific parameter, including potential bioaccumulation effects. Additionally, WET testing may be used to determine the overall effect of an effluent on aquatic life. This would include the synergistic effects of multiple parameters together, or the effects of parameters in the effluent that may not have numeric standards or cannot be measured by current analytical methods.

2.1.1 Components of Aquatic Life Criteria

The magnitude component for aquatic life criteria is divided into two different expressions, acute and chronic. The acute magnitude is typically derived using 48 to 96-hour toxicity tests that measure lethality or immobilization. For a given parameter, the allowable concentration for the acute criteria is typically higher than the chronic criteria, but for exposure over a much shorter duration. Chronic effects are determined from 28-day (or longer) toxicity tests that evaluate lethality, growth or reproduction, and bioaccumulation. The allowable concentrations for chronic effects are typically much lower than for acute effects, but for a longer duration.

For duration, the TSD recommends an averaging period of 1 hour for acute criteria and 4 days for chronic criteria. These durations are based on the fastest acting toxicants (e.g., chlorine) and are, therefore, protective of toxicants that are slower to show an effect (EPA, TSD 1991). The duration component is implemented directly if dynamic modeling is used to develop a WLA or WQBEL. This component is also used in the development of design flows for steady state modeling.

In the TSD, USEPA recommends an allowable frequency of exceedance of once in three years for aquatic life protection. This was established in USEPA's report on the Guidelines for Deriving Criteria (EPA, 1985). This frequency was set to coincide with the applicable low flow criteria of 7Q10, or the 7-day low flow that would recur once in 10 years. The basis of this is that a slight excursion over the acute or chronic criteria would have little effect or require little time for recovery. Like the duration component, the frequency component can be used directly in dynamic modeling of WLAs and WQBELs, and is also used in the development of design flows for steady state approaches.

These components are used in the setting of water quality standards (WQS) and in the development of permit limitations. Limitations are often expressed in permits as maximum daily limits and weekly or monthly averages. Statistical approaches described in the TSD are used to translate between the duration components of the criteria (e.g., 1-hour for acute, 4-day for chronic) and the averaging periods of the effluent limitations. These methods are discussed in greater detail in Section 4.

2.1.2 Direct Toxicity of Nutrient Species

As mentioned earlier in this section, specific nutrient species can cause toxicity to aquatic life. Ammonia may cause lethality, growth, or reproductive effects at relatively low concentrations (depending on factors such as pH and temperature), and the criteria are based on magnitude, duration, and frequency as outlined above. The toxic effects of ammonia may include damage to gills, reduction in oxygen carrying capacity, depletion of adenosine triphosphate (ATP), and may also effect the liver and kidneys (USEPA, 2013a). Because of these effects, ammonia is treated as a toxic parameter and standards are developed in accordance with a one-hour exposure level (acute) and a thirty-day exposure level (chronic). This protects aquatic life by ensuring that the short term instream concentrations do not cause lethality and that the longer term exposure does not cause lethality, growth, or reproductive effects.

Nitrate and nitrite can also cause toxicity to vertebrates and invertebrates mainly due to oxygen delivery (Camargo and others, 2005; Camargo and Alonso, 2006). Although nitrite criteria exist for aquatic life protection, the most stringent nitrate criteria are for protection of human health for water supply-type designated uses. As of 1998, seventeen states had no specified criteria for nitrates or nitrites. The other states had some combination of narrative and/or quantitative criteria which may be based on just drinking water protection or other criteria. Nine states used only the nitrate criteria of 10 mg/L (acute standard) for the protection of domestic drinking water supplies (USEPA, 1998). There is little guidance for the protection of aquatic life from nitrate toxicity, as most information centers on ammonia and nitrite which are more toxic. Depending on species, acute endpoints for nitrate toxicity for aquatic life range from 100 mg/L to 1900 mg/L (Monson, 2010) while some studies have recommended chronic criteria as low as 2 mg/L (Camargo and others, 2005).

Nutrient standards for total nitrogen (TN) and total phosphorus (TP) are set for protection of aesthetic or aquatic life from eutrophication impacts, which do not follow this same type of one-hour or four-day timeframe. Excessive algae growth does not occur or manifest effects as quickly as toxic impacts to aquatic life. In fact, eutrophication may not occur in waters that have high concentrations of TN and TP if other conditions are not present (Yang, 2008). Because eutrophication causes indirect effects, more time is needed, at optimal conditions, for excessive growth of algae and the associated indirect impacts to occur.

2.1.3 Eutrophic Effects on Aquatic Life in Streams

Eutrophic effects in streams can be expressed as the presence of elevated macrophytes, attached periphyton in wadeable streams, as well as phytoplankton in larger and deeper rivers. Nutrient concentration is only one factor that affects the algal growth in streams, and growth rates can often be controlled by other factors such as light availability, hydraulics, and substrate availability. For this reason, stream responses to nutrient loading require more a complex evaluation than that of toxics parameters.

Excessive algae blooms can result in aesthetic (e.g., excess algae, increased turbidity, odor) or aquatic life impacts (e.g., low oxygen levels, increased pH, lower light levels) for fish and macroinvertebrate species (Yuan, 2010) and certain types of algae can produce toxic compounds that are a health threat to wild or domestic animals and humans. Numerous studies have looked at the effects of nutrient enrichment on macroinvertebrate species. In some cases enrichment has led to increased populations or biomass and in others decreased populations or biomass of macroinvertebrates (Miltner, 2010). Other studies have shown shifts in the types of macroinvertebrates present at higher algal densities (Dudley and others, 2002; Ortiz and Puig, 2007; Gafner and Robinson, 2007; Singer and Battin, 2007). These effects may be the result of a single change or combination of changes in food availability, habitat quality, oxygen levels, light penetration, water chemistry, and specie interactions.

Low oxygen levels are detrimental to aquatic life and can cause fish kills. Higher biomass accumulation can result in large diurnal swings in oxygen levels which also may lead to short episodes of hypoxia (Sabater and others, 2000). Additionally, both macroinvertebrate condition and sensitive fish taxa may decline with large oxygen swings (Miltner, 2010). Low oxygen levels also impact fish growth and reproduction, leading to reduced fish taxa populations (Arend and others, 2011).

Although pH changes occur based on photosynthesis and respiration, large swings in pH may have both direct and indirect effects on aquatic life. Outside of the range of 6.5 to 9.0, pH may cause damage to gills, eyes, and skin. Higher pH levels increase the toxicity of ammonia while lower pH levels may increase metal toxicity.

The growth of macrophytes can reduce flow rates, thereby increasing sedimentation rates, reducing turbidity, and allowing more light penetration, which can further stimulate growth of macrophytes (Madsen and others, 2001). Increased sedimentation and reduced turbidity may result in fewer habitats for macroinvertebrates as there is less space between rocks (May, 2010). In addition, sedimentation and additional macrophyte growth can reduce spawning areas for aquatic fish (International Symposium on Eutrophication, 1969). Increased light may also increase the growth rate of algae. Algal blooms can increase turbidity and limit light penetration which will reduce growth and cause the die-off of plants; however, when these algal blooms die off decomposition can further deplete oxygen levels (Chislock, 2013). Excess turbidity may also lead to fewer photosynthetic organisms and a reduced rate of oxygen production (NERR, 2014). This may also result in a reduction of available food sources for many invertebrates or fish species.

Although algae can grow rapidly, with growth rates up to 21 percent per day (Taylor and others, 2001), maximum growth rates require a combination of conditions to occur, and the presence of nutrients may or may not trigger excessive growths depending on other factors such as flow velocity or temperature (Yang, 2008). Unlike toxic parameters, these effects are indirectly related to the presence of nutrients in stream and will not occur if concentrations of nutrients are above any threshold levels in a one-hour or four-day timeframe. Conversely, standards or limitations for toxic parameters are relatively higher for the one-hour (acute) effects and relatively lower for the four-day (chronic) effects. Because algae blooms are typically limited to warm months, the duration component for nutrients should differ from that of toxics.

As nutrient (and other factors) based responses are slower to occur than those of toxic substances and are not subject to the same dose-response evaluations as toxics, temporary exceedances of nutrient criteria are unlikely to invoke an acute-type response in streams. For streams, the seasonal timing and longer term consistent concentrations are more important to nutrient impacts than short term concentrations or even total loading of nutrients to the system.

In addition, the response of eutrophication to critical flow conditions may vary and may be stream dependent. For constituents associated with particulates, concentrations generally increase with increasing streamflow, while for other constituents higher flows lead to lower concentrations based on available dilution (USGS, 2008). For phosphorus, higher flows tend to result in higher concentrations (Sadek and others, 2005). Some studies have concluded that greater nitrogen concentrations generally coincide with less flow (Scott and others, 2010), while others have shown that total nitrogen concentrations tend to increase with flow (MCD, 2011). Critical flow conditions are likely to be site-specific and take into account factors such as groundwater base flow, groundwater chemistry, soil chemistry, amount of particulate matter, surface runoff, seasonal changes in flow patterns, and land use.

2.1.4 Eutrophic Effects on Fisheries in Lakes and Reservoirs

Lakes and reservoirs can respond to nutrient loading in very different manners than rivers and streams, largely due to differences in morphometry and hydraulic residence time. Deeper water bodies are much more likely to vertically stratify, and therefore experience hypolimnetic hypoxia. Hydraulic retention time can influence biogeochemical cycling, thermal stratification, and particle settling; therefore, influencing nutrient availability, turbidity, mixing, and transport to downstream receptors. Due to long hydraulic retention times and internal nutrient recycling, lakes and reservoirs can respond to nutrient loading on an annual or seasonal basis, rather than short-term basis.

Internal and external sediment loading may have long-term eutrophication implications in the environment. When algae die-off occurs at the end of the growing season, the algae cells that have a tendency to absorb and concentrate mineral nutrients settle to the bottom of the lake. In addition, phosphorus adsorbed to suspended solids will settle to the bottom of the lake. These become internal sources for nutrients during periods of low external loading, leading to eutrophic effects long after external nutrient loading is reduced (Sondergaard, 2003). The significance of internal loading can depend on the flushing rate, the loading history, and the sediment’s chemical characteristics and can be redox or microbial controlled. Internal loading has led to summer phosphorus concentrations exceeding winter concentrations by a factor of 2 to 3 times (Sondergaard, 2003).

In addition to internal nutrient releases, organic material accumulating at the bottom of lakes from algae and plant die-off undergoes microbial decomposition resulting in depleted oxygen and greenhouse gas or toxic gas (methane and hydrogen sulfide) production. Depleted oxygen levels have a significant effect on fish and invertebrate habitat and population and can also lead to sub-lethal effects (altered fish physiological processes, spatial migration, disrupted predator-prey interactions, impacts to fish growth and reproduction, and changes in community composition) (Arend and others, 2011). Coldwater fish (e.g., trout and salmonids) and highly benthic species are most sensitive to these shifts in the environment (Jones-Lee and Lee, 2005; Arend and others, 2011), as an oxygen depleted environment drives these species into warmer waters ultimately leading to population decreases. A study of Lake Erie found oxygen depleted environments can occur seasonally from mid-July to mid-October (Arend and others, 2011); however, oxygen depletions in other lakes may be shorter depending on hydraulic residence time and geographic location.

As lakes become more turbid with a high abundance of phytoplankton, fewer submerged macrophytes and more planktivorous fish have been observed (Jeppesen, 1990; Jeppesen, 2005). A decrease in submerged vegetation reduces the habitat preferred by some invertebrate and fish taxa, including the early life stages of many fish species (Jones-Lee and Lee, 2005). Additionally, turbid conditions and the resulting low light penetration may favor the development of cyanobacteria (blue green algae) that can also be problematic (Chorus and Bartram, 1999). Certain types of cyanobacteria produce toxins that can adversely affect both aquatic life and human health. This topic is discussed more in the human health section below.

Despite the detrimental effects eutrophication can have on some uses, nutrient enrichment to certain levels can actually be beneficial to warmwater fisheries. Low to moderate levels of eutrophication can be favorable and promote fish population increases due to additional food supply and macrophyte-dominated habitat, increasing prey, and enhancing reproductive success (Willemsen, 1980). In these circumstances, the desired level of eutrophication can be a trade-off between the desire to promote the warmwater fishery and the desire to promote clearer water for recreational aesthetics. Higher levels of eutrophication can be detrimental even to warmwater fisheries. Although total fish biomass might still increase, the quality of the fishery can be degraded by an overabundance of unwanted species such as carp (Tammi and others, 1999).

Similar to rivers, several factors are involved in a lake's response to nutrient loading leading to a more complex threshold evaluation than that of toxic parameters. Again, short term exceedances (one-hour acute) of nutrient criteria are unlikely to invoke a direct response in algae growth (Yang, 2008; Taylor and others, 2001). Longer term exceedances may invoke a response depending on the actual duration and timing (seasonality). Due to the accumulation of sediment, recycling of nutrients, and the residence time of water in lakes (as opposed to the flow-through nature of streams), lakes may be more susceptible to time-averaged loading of nutrients rather than short-term concentrations (Jones-Lee and Lee, 2005; Arend and others, 2011). Anoxic conditions—and especially iron-reducing conditions—facilitate the release of phosphorus from bottom sediments, and thus can cause a positive feedback loop of eutrophication. For at least one major reservoir, it has been demonstrated that maintaining higher nitrate levels actually benefited water quality by reducing the release of phosphorus and ammonia from sediments (Cubas and others, 2014).

Critical condition occurrence may be less apparent in lakes in comparison to streams, because high flows may increase nutrient loading but may also have an offset effect due to increased flushing rates (i.e., decreased hydraulic residence times). High flushing rates limit the development of phytoplankton biomass and subsequently eutrophication effects within lakes (Virginia Water Resources Research Center, 2007). Therefore, critical conditions as linked to wet or dry years are likely lake specific, where dry years may be critical for riverine segments and wet years may be more critical for lacustrine segments.

2.1.5 Eutrophic Effects in Estuaries

As large and complex environments, estuaries and coastal waters can respond to nutrient inputs in many different ways. The most common problems attributed to eutrophication in estuaries include: reduced dissolved oxygen, reduced water clarity, loss of submerged aquatic vegetation, harmful phytoplankton blooms, and nuisance growths of macroalgae (Bricker and others, 2007). The susceptibility of an estuary to these effects is a function of various system-specific characteristics such as hydraulic retention time, stratification, top-down controls, and even tidal amplitude (Cloern, 2001). Large zones of hypoxia/anoxia (i.e., "dead zones") are a major stressor to many types of marine life including fin fish and benthic macroinvertebrates (Diaz and Rosenburg, 2008).

These zones develop in response to nutrient loading over seasonal, annual, or even multi-annual time scales. Aquatic life impacts of hypoxia include mortality, reduced growth, and inability to reach favored habitats (Breitburg, 2002). Nutrients can contribute to submerged aquatic vegetation (SAV) loss both by increasing the algal biomass in the water column and supporting epiphytic growth directly on the SAV (Twilley and others, 1985). Loss of SAV can impact other organisms that depend on seagrass beds for food or shelter—particularly larval stages that use seagrass beds as a nursery habitat. However, in areas with high inorganic suspended solids in the water column, nutrients/chlorophyll *a* may be only a minor contributor to light attenuation.

Eutrophication contributes to potentially harmful algal blooms (HABs) in the estuarine environment, such as red or mahogany tides (Heisler and others, 2008). Cyanobacteria are the most common HAB-former in tidal freshwater areas, where most HABs in more saline waters consist of dinoflagellates. A small proportion of bloom-forming phytoplankton are toxic to aquatic life and severe blooms of toxic strains can cause direct mortality of zooplankton, shellfish, fish, or even marine mammals (Landsberg, 2002). Non-toxic HABs can also cause indirect effects such as physical damage to gill structures and reduced feeding rates. Although nutrients are one factor that contributes to HAB formation, HABs such as red tides form in response to a complex set of physical and chemical stimuli that cannot be easily predicted. The global spreading of dinoflagellate cysts in ballast water is another factor that is believed to have contributed to an increased HAB occurrence (Hallegraef, 1998).

2.2 Public Water Supply

Public water supply is one of the chief designated uses of our nation's waters, and so merits consideration with regard to how nutrient impacts compare/contrast with toxics impacts. The TSD approach for protection of human health relies on the exposure routes of fish consumption and drinking water as well as a specified risk level for carcinogens. Unlike aquatic life criteria, human health criteria for toxics relies solely on protection against long term or chronic health effects. For carcinogens, the duration is for a lifetime of exposure, deemed 70 years. For noncarcinogens, duration may be determined by whether effects are based on age or gender-specific developmental issues, lifetime exposure, or organoleptic effects which may not be duration based. Bioaccumulation issues are also taken into account. In general, criteria correspond to a lifetime of consumption of so many grams of fish and so many liters of water per day, for 70 years, and an associated cancer risk. States may use different grams of fish consumption based on population averages, and different cancer risk thresholds may be used for different parameters.

With the exception of higher nitrate levels in drinking water, nutrient effects on human health are indirect, and center around the production of toxic chemicals from certain types of algae. Cyanobacteria, or blue-green algae, may produce cyanotoxins, the most widespread of which are the peptide toxins called microcystins (EPA 2012). Cyanotoxins may affect the nervous system, liver, or skin and may cause a wide range of symptoms and, in extreme cases, may cause death. Anatoxin-a is another toxin that may be produced by cyanobacteria and may cause paralysis followed by asphyxiation (Stone and Bress, 2006). Formation of cyanobacteria blooms occur most often in the late summer but some strains can occur early in the summer (EPA, 2012).

Drinking water facilities as well as industrial and agricultural users may all be impacted by the eutrophication of streams and lakes. Excessive algae can clog water treatment systems making it more expensive to treat drinking water. This may also lead to the use of additional chemicals, increase the frequency of filter back-flushing, and require additional time for settling in order to improve the quality of drinking water (Nordin, 1985, USEPA River and Streams Guidance). Additionally, carcinogenic disinfection by-products (e.g., trihalomethanes) may be formed by disinfection processes in water treatment facilities when the raw water supply has certain organic compounds present, which may be formed by the decomposition of organic matter including algae (EPA, 2000).

Because nutrients themselves are not toxic to humans at the levels being regulated, the magnitude and duration components of the TSD for human health would not apply directly to nutrients. Procedures for deriving criteria for individual algal toxins could be more directly analogous to those for deriving toxics criteria. However, the derivation of nutrient permit limits would have to consider the more complex linkages between nutrients, algae, and toxin production.

2.3 Recreation

Many states' water quality standards state that waters should be "fishable and swimmable", and so recreation is an imported designated uses for consideration of the differences between nutrient impacts and those of other pollutants. Similar to the human health section above, some recreational standards are based on impacts to humans. Recreational-based water quality standards have been established for bacteria and toxic parameters to which swimmers/boaters might be explored through dermal absorption and incidental ingestion (EPA TSD). As with aquatic life and public water supply uses, recreational uses would not generally be directly affected by nutrient concentrations. Rather, the nature of recreational impacts would depend on secondary responses to nutrients that could be water body-specific.

Excessive algae growth can cause issues in regards to boating, swimming, fishing, and aesthetics. It can increase the risk of drowning by entanglement in plant growth or limiting the ability to see underwater hazards. The potential for odors as well as the look and feel of water associated with excessive algae growth can make it unattractive for swimming or other recreational uses. Based on a user perception study, a threshold value for chlorophyll *a* of 150 mg/m^2 for attached growth has been proposed (USEPA, 2000). However, chlorophyll *a* levels acceptable to the public can vary greatly based on regional expectations (Heiskary and Walker, 1988).

Although some indirect effects of nutrients in water bodies can lead to conditions that may alter recreational fisheries. Numerous studies (e.g., Oglesby, 1977; Hanson and Leggett, 1982). have shown that total fish yield in reservoirs tends to increase with algal production. Drops in algae production may have a negative effect on warmwater fisheries. Declines in sport fish populations have been found to occur in systems with reduced nutrient inputs (Axler and others, 1988; Ney, 1996). However, eutrophication also decreases dissolved oxygen concentrations in the hypolimnion, and so the impacts of nutrients on fisheries represent a trade-off between higher food supply and lower habitat availability, and coldwater fisheries are more prone to adverse impacts than warmwater fisheries. Under highly eutrophic conditions, reservoirs can become dominated by less desirable fish such as carp (Lee and Jones, 2012).

It is the indirect nature of nutrients in these systems that may cause these types of issues. Using the aquatic life or human health methods in the TSD to set limitations for nutrients is not appropriate for protection as it may be under- or over-protective of various uses. As different water bodies may react differently to a certain level of nutrients, the magnitude, duration, and frequency components need to be tailored to each particular water body. As different water bodies may react differently to nutrients, a one size fits all approach to nutrient standards does not work.

2.4 Other Uses

In addition to the aquatic life, human health, and recreation impacts noted above, excessive algae growth can also impact other uses. Depending on the specific use, industrial users of water may also have to treat prior to use of water leading to increased costs and the same issues faced by drinking water suppliers. Excessive algae growth may have an impact to agricultural irrigation where algae can clog nozzles, pipes, pumps, or drip systems and chemistry changes such as pH can effect crop growth. Ranchers may face issues where cyanobacteria and associated toxins may be harmful to livestock.

2.5 Conclusions

Based on the literature surrounding nutrients and eutrophication, different water systems react differently to nutrient inputs. In some systems, small amounts of nutrients may cause excessive algae blooms, whereas in other systems greater amounts may be needed to have the same effect or an effect may not be seen as there are other limiting factors. Some systems may only need controls on phosphorus, others only nitrogen and some may be co-limited and require control of both. In fact, too little nutrients may have adverse effects on fisheries. Lakes and reservoirs may react differently than streams and rivers and may be more suited to load-based controls rather than concentration-based controls as a result of nutrient cycling in the lake system. The amount of flow in a river or the velocity of flow may have more of an effect on algae growth than the presence of nutrients. In other words, there is no "one size fits all" approach that will protect against the adverse effects of excessive algae growth.

The magnitude, duration, and frequency of nutrient levels differs from that of the TSD approach to controlling toxics for the protection of aquatic life or human health. The TSD approach centers around a dose-response methodology of determining the concentrations of a parameter that will have an adverse effect on aquatic life and human health. The dose and response in the case of nutrients is not a direct relationship and is heavily influenced by both chemical and physical factors that drive the actual impact on the use.

Also, unlike many toxics, timing of nutrient availability may also be a factor for some water bodies in that algae growth typically occurs in the summer months. Some states have attempted to address the timing issue by using annual averaging periods in permits for nitrogen and phosphorus criteria. This may be appropriate for some systems such as lakes where nutrients may be recycled throughout the system at different times of year, or for streams that directly feed a lake system. However, this may be less appropriate for rivers and streams where the potential growing season may be extremely limited or where the other necessary conditions are lacking for algae growth during part of the year. Nutrient criteria may best be developed for specific water bodies based on site-specific information and studies that take into account the conditions unique to that system rather than assuming that they fit a dose-response model.

Section 3

Nutrient Characteristics of Wastewater Effluents

Wastewater treatment plants play an important role in controlling nutrient discharges to our water bodies. There are a variety of treatment approaches available that can achieve reliably low effluent limits if appropriate averaging periods are applied. Evolving and emerging technologies are not necessarily causing major reductions in the attainable effluent concentrations, but are improving costs and sustainability of point source nutrient controls. Regardless of whether effluent limits are water quality-based or technology-based, nutrient permitting requires an adequate understanding of the nutrient characteristics of wastewater effluents. Statistical procedures for deriving nutrient WQBELs require specification or estimation of the distribution of nutrient concentrations in effluent, including the frequency distribution and factors that describe effluent variability (e.g., coefficient of variation). To avoid setting unachievable permit limits, it is necessary to understand the role of effluent variability and averaging periods on the attainability of different levels of nutrient control. This section addresses these issues by summarizing recent USEPA and WERF research on the nutrient-related characteristics of effluents.

The recent research reported here in shows that significant statistical variability is a characteristic of all nutrient removal plants and that this variability has to be considered in both identifying appropriate technologies in engineering the plants as well as determining appropriate limits in a regulatory setting process. The research shows that when seeking very low effluent nutrient levels, even exemplary plants may experience violations with limits that have an inappropriate or unrealistic averaging period, no matter how much effort, cost and energy is expended. Finding proper statistical bases for regulation that are both protective of the environment and are technologically achievable should be the goal.

Regulated nutrients in wastewater effluents for prevention of water quality impacts from eutrophication are typically expressed in terms of total phosphorus or total nitrogen although, for prevention of human health impacts, total inorganic nitrogen is often regulated instead of total nitrogen. Total inorganic nitrogen includes nitrite, nitrate, and ammonia but excludes the portion of total nitrogen associated with organic nitrogen. Statistically comparing effluent quality for various permit averaging periods allows performance to be linked to reliability for various treatment processes. It has been found that performance and reliability depends on the combination of treatment processes employed and their design (e.g., Kang et al., 2008; Bott and Parker, 2011).

3.1 Statistical Distribution of Nutrient Concentrations

This section describes the TSD's approach for characterizing the frequency distribution of effluent concentrations, and compares TSD assumptions to frequency distributions derived from actual nutrient monitoring data from facilities implementing nutrient control treatment. This section also discusses the implications of effluent variability and averaging period on the expected frequency of permit limit violations.

3.1.1 The TSD Approach

The TSD was developed in 1991 and its statistical bases have not been reexamined as to appropriateness for other types of pollutants until recently. The TSD approach (USEPA, 1991) to handling effluent statistical distributions in permit setting is based on several concepts, some of which are unique to toxics regulation.

1. _Default assumptions_. In many permitting situations, there is limited ability to characterize the statistical characteristics of effluent concentrations (see Section 3.3.2 of the TSD). In these circumstances, the TSD recommends using the default assumption that concentrations are lognormally distributed with a coefficient of variation (CV) of 0.6. Maximum values of this assumed distribution, even if never observed in the effluent, are used to determine whether the discharge has a reasonable potential to cause an exceedance of water quality criteria in the receiving water. The TSD method acknowledges that larger data sets lead to increased statistical certainty in regulation. If there is a larger data set to characterize effluent variability for the chemical of concern, it can be used to define a statistical distribution and coefficient of variation in lieu of using the default assumptions (Section 5.5.2 of the TSD).

2. _Limit expression_: The TSD follows the framework of the NPDES regulations as quoted in Section 5.2.3:

   > The NPDES regulations at 40 CFR122.45 (d) require that all permit limits be expressed, *unless impracticable,* as both average monthly and maximum daily values for all discharges other than POTWs and as average weekly and average monthly limits for POTWs. The maximum daily permit limit (MDL) is the highest allowable discharge measured during a calendar day. The *average monthly* permit limit (AML) is the highest allowable value of the average of daily discharges obtained over a calendar month. The *average weekly* permit limit is the highest allowable value for the average of daily discharges obtained over a calendar week [emphasis added].

   The primary motivation for the short averaging periods is to protect aquatic organisms from toxic impacts that manifest over short time frames (hours to a few days). As discussed in other sections of this document, longer averaging periods are usually more appropriate for nutrients (e.g. monthly, seasonal, or annual timeframes).

   USEPA's own analysis of nutrient data in municipal effluents (Kang and others, 2008) evaluated only a single year of data from various nutrient treatment levels from a facility, (e.g., low (TN <3) and mid-level (TN 3 to 6 mg/L) nitrogen removal and low-level (TP < 0.1) phosphorus removal). Both U.S. and Canadian plants were evaluated. The seven U.S. plants surveyed by USEPA (Kang and others, 2008) included plants with the following averaging periods in their permits:

   a) Average monthly and average weekly TP limits.
   b) Maximum day TP mass discharge limits.
   c) Annual TN load limits and quarterly TP concentration limits.
   d) Annual TN load limits and quarterly TP concentration limits.
   e) Annual average, monthly average, and weekly average TN and TP concentration limits.
   f) Annual average, monthly average, weekly average, and daily maximum TN and TP concentration limits.
   g) Annual average, monthly average, and weekly average TN and TP concentration limits.

It is notable that only one of the permits for nutrients follows the TSD framework of monthly and weekly average concentrations while others focus on periods as long as annual or quarterly. Also, only one had a maximum daily permit value for nutrients. In contrast to permit limits for the U.S. plants, the surveyed permit limits for Canadian plants were expressed as percentile values. As discussed by Kang and others (2008), permit limits that are expressed as a value with an averaging period can also be expressed as a percentile (Table 3-1). In this manner, the U.S. and Canadian methods for expressing permit limits are related ways of quantifying the desired statistical performance of a plant.

Table 3-1. Statistical Comparison of U.S. and Canadian Permit Limits (from Kang and others, 2008)

Statistical Percentile	U.S. Term – averaging of maximum period	Canadian Term – percent less than value
50	Annual average	50th percentile
80	Maximum quarter	
90		90th percentile
92.3	Maximum month	
98.1	Maximum week	
99		99th percentile

3.1.2 WERF's Evaluation of Statistical Approaches for Regulation of Nutrients in Effluents

The cited USEPA work on nutrient removal performance (Kang and others, 2008) inspired the Water Environment Research Foundation (WERF) to examine the reliability and performance of a number of exemplary nutrient removal plants across North America (Bott and Parker, 2011). A specific goal of the WERF project was to document how the length of averaging periods affects the attainability of low permit limits. The WERF work was based on the premise that measurable nutrient impacts normally occur on a longer term basis – usually longer than the monthly or weekly basis used for conventional pollutants (and suggested for toxics in the TSD). The work is relevant to this TSD review, in that the TSD document is oriented towards preventing discharge of high concentrations of toxic chemicals impacting organisms over short exposure periods, a situation which does not normally apply to nutrient discharges.

The WERF project involved examination of 3 years of monitoring data with low levels of nutrient reduction. Across North America, a total of 19 nutrient removal treatment plants had the required data. The project used a longer period of record than the single year used by the earlier USEPA work (Kang and others, 2008) to help determine the long term performance of low level nutrient removal technologies. Additionally, 3 plants were included that have maximum daily limits for ammonia, derived from toxicity concerns. The project approach was to develop a database for assessing whether specific effluent limits could be reliably met. WERF used 36 months of data to represent a range of climatic, loading, and operational conditions in an effort to better represent a 60-month NPDES permit period. Many of the plants with more advanced treatment schemes had not been in operation for a full NPDES permit period and would have been excluded if the full 60-month period of available data was the threshold for them to be included.

The work reported by WERF represents the most comprehensive statistical evaluation completed to date of North American plants facing a variety of climatic conditions achieving low nutrient concentrations.

The investigation was run by a steering committee of WEF and WERF representatives and, while the USEPA had no funding involvement, they provided a steering committee member because of their research and regulatory interests in the subject area. Additionally, as for any WERF project, a WERF technical review committee provided QC review for the work.

Bott and Parker (2011) should be examined for full details of the WERF project, but the statistical evaluation procedure for processing the data can be summarized in a small number of steps. To provide a consistent approach between plants, all values reported as zero were converted to half the minimum detection limit (MDL). This allowed transformation of effluent numbers to their log form for plotting and analysis as the next step (non-zero numbers are required). Ranking the data and associating them with a probability was done according to the commonly used Weibull probability formula:

$$P = \frac{\text{rank}}{n+1} \quad (3.1\text{-}1)$$

Where: P = probability and n = number of data points in the set.

The values were plotted after the probabilities were calculated as their log values for plotting on the y-axis. Summary statistics were calculated for the full data set including the arithmetic average (mean), geometric mean, standard deviation, coefficient of variance (CV), skew, minimum, and maximum. Percentile statistics were also calculated from the same data set at certain values (the 3.84th, 50th, 90th, 95th, and 99th values). These percentiles are referred to as the probability that a value is less than or equal to the stated concentration.

Data were summarized using 30-day rolling averages, monthly averages, and 12-month rolling averages of the monthly averages (or annual averages). Thus, each data set yielded four sets of summary statistics and probability plots (raw daily data, 30-day rolling average, monthly average, and annual average). Plant data consisted of multiple data sets. The most data sets were for nitrogen removal plants, where statistical data were reported for TN: ammonia nitrogen, nitrite and nitrate nitrogen (or NOx-N), and organic nitrogen. For phosphorus, TP and orthophosphate as P were evaluated.

“Best fit” lognormal curves were plotted on the curves vs. the actual data. The report adopted two statistical evaluation procedures. The first used equations developed by Niku et al. (1979) and extended by Olivera and Sperling (2008) based on lognormal distributions to calculate reliability in meeting requirements. For the second procedure, probability values were taken directly from the data without the assumption that the data were lognormally distributed. Examples of the two different approaches are shown on Figure 3-1, where the raw daily TP data are analyzed for the Iowa Hill plant. On the figure, the 90th percentile value is identified from the data itself, whereas the reliability of meeting a TP treatment objective of 0.05 mg/L identified from the lognormal fit line is 95.7 percent. The reliability could also be identified from the data and, in this case, the reliability would fall slightly below 95.7 percent.

Technology evaluations in the WERF study relied on the observed data distributions, expressed on a probability basis, because the fitted lognormal curves tended to depart from the data in the region of most interest (high nutrient concentrations at low averaging periods). The focus here was on the daily, rolling 30-day average, monthly, and annual averages at the 50, 90, and 95 percentile values. Given that NPDES permits represent five years of data, the percentile values can be converted to expected violations per permit period. Note that these percentiles differ from those represented in USEPA’s prior work (Table 3-1), because the USEPA values of Table 3-1 were based on a single year rather than a five-year permit term. For example, in Table 3-1, a 92.3 percentile value means that, statistically, one monthly exceedence per year would be expected.

Taking a five-year permit duration perspective, even a 95 percentile value would represent 3 months of violation per permit period. Even for this broadened permit period perspective, the WERF investigators noted that projecting over as long as 2 or 4 permit periods (ten or 20 years) may not be accurate given that 36 months of data do not reflect all that a plant might experience over multiple permit periods.

Because CVs are an intermediate step in the fitting of the distributions, the WERF investigators did not separately report them in their report. However, CVs were subject to wide variability and resulted in values both above and below the 0.6 value used in the TSD. There were insufficient data to generalize about how the CV would change with treatment process. For this reason, the TSD's default assumption of a CV of 0.6 could neither be refuted nor confirmed. Rather, the results point to the importance of determining CVs from facility-specific data.

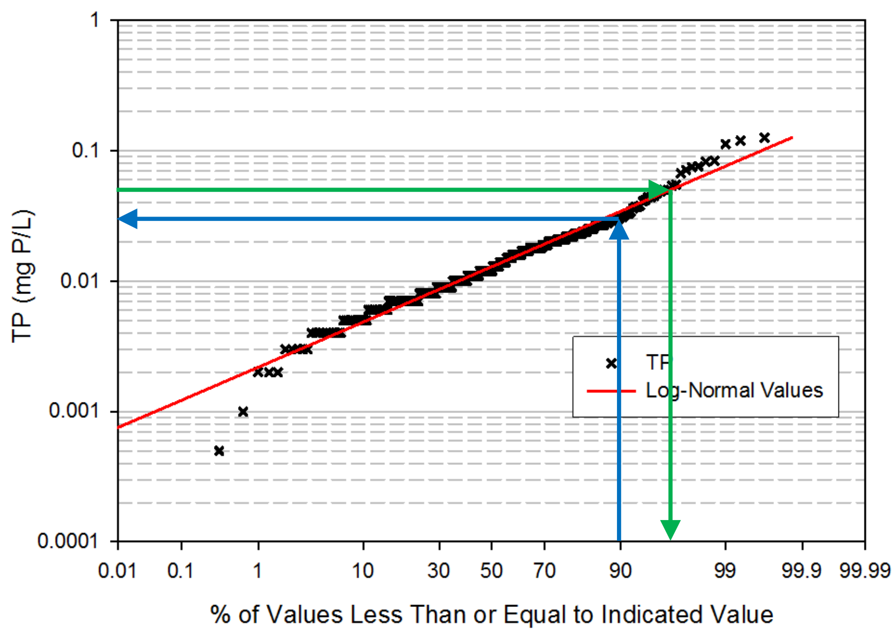


Figure 3-1. Probability Plot for Daily TP Data for the Iowa Hill WRF, Breckenridge, CO (Bott and Parker, 2011).

Table 3-2. Number of Exceedances Per Five-Year NPDES Permit Period for Daily, Monthly, and Annual Average Permits for Given Percentile Values (after Bott and Parker, 2011)

Percentile Less than Stated Concentration	Expected Number of Daily Average Exceedences (with Daily Sampling)	Expected Number of Monthly Average Exceedeences	Expected Number of Annual Average Exceedences
Total reporting events in 5 years	1,826	60	5
50	912	30	2.5
90	183	6	0.5 (or 1 per 2 permit periods)*
95	91	3	0.25 (or 1 per 4 permit periods)*
99	18	0.6 (or 1 per 2 permit periods)*	0.05 (or 1 per 20 permit periods)*

Note: * These percentile values can only be calculated assuming the longer periods are adequately represented by 36 months of data.

Example plots for all the nitrogen species for the Truckee Meadows Water Reclamation Facility are shown on Figure 3-2. It can be see that, of the nitrogen species, the species departing most from the lognormal distributions are the values for ammonia, because these are impacted by values below the MDL. Even for the monthly ammonia plots, there was a region at low concentrations where the same value was frequently reported. This was a common characteristic when examining ammonia statistics in nitrifying plants.

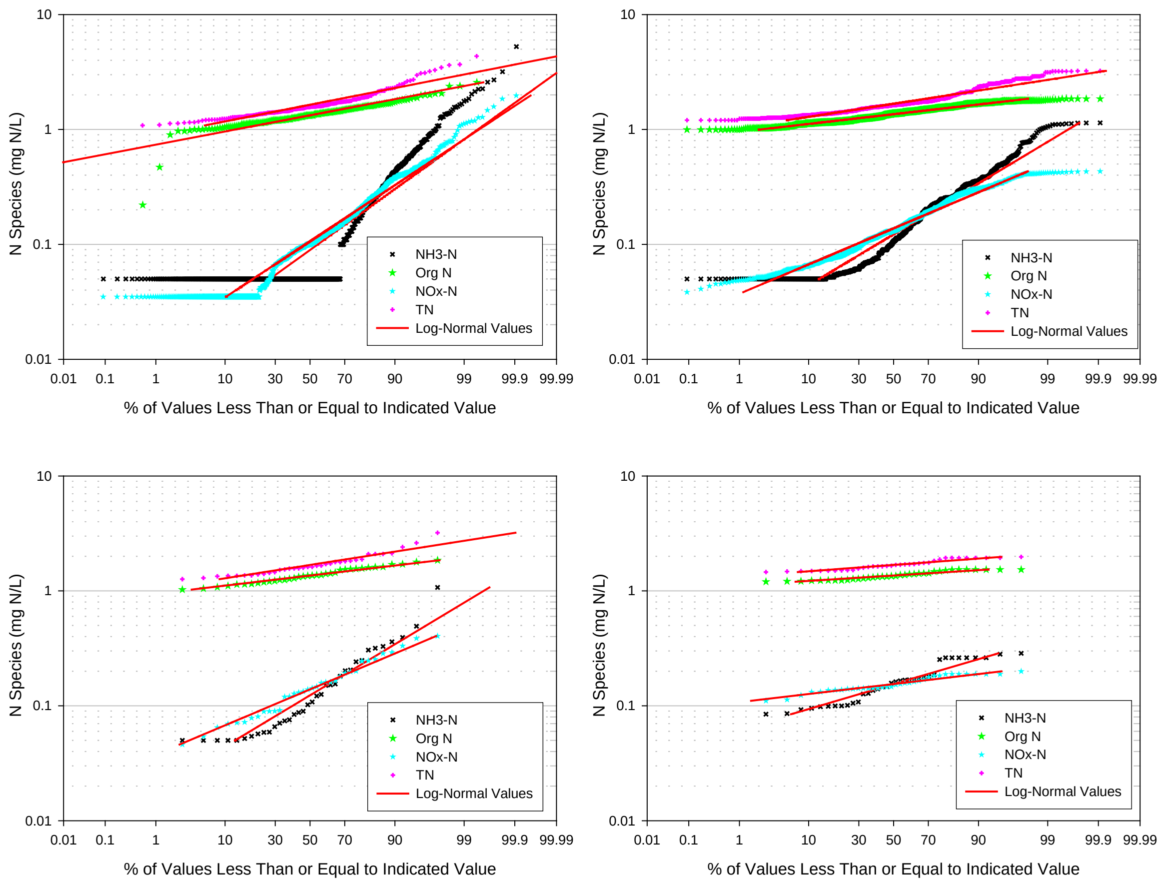


Figure 3-2-. Probability Plots for TMWRF – (A) Daily Data; (B) 30-day Rolling Average; (C) Monthly Averages; (D) Annual Average (Bott and Parker, 2011).

Using the probability summary for TN on Figure 3-2 as an example, one can use the probability plots to determine the probability of attaining effluent limits for various nutrient targets and timeframes. For example, following are the number of violations that would be expected over a 5-year period term, if the TN limit was 3 mg/L:

- If a daily limit: 70 daily violations
- If a monthly limit: 2 monthly violations
- If an annual limit: no violations

This demonstrates a general phenomenon observed in the WERF work, that for the same numerical effluent requirement, the longer the averaging period, the lower the probability that an effluent violation will occur. Longer averaging periods are usually appropriate for nutrients due to the manner in which nutrient affect receiving waters, and the manner in which nutrient criteria are typically derived.

If nutrient limits were derived using observed probability distributions such as those discussed in this section, it would not be necessary to make TSD-like assumptions regarding the data distribution or the CV, which can result in overly stringent nutrient limitations that are not required to protect the receiving water from nutrient impacts (e.g., the C.V. of 0.6 as used in Appendix E of the TSD may not be representative of the variability of nutrient data in effluents).

This approach would require a relatively long (e.g., 3-year) period of monitoring data. Unless the limits were based on existing water quality conditions, this approach would also require an adjustment in the magnitude of the concentrations, in order to meet WLAs. Empirical alternatives for deriving WQBELs are discussed further in section 4.3.3.

3.2 Attainable Effluent Quality

The focus of many NPDES permits for secondary treatment was based on the capability of the facility to meet a technology-based standard for conventional pollutants. In fact, the use of monthly and weekly averages for conventional pollutants of 30 mg/L and 45 mg/L BOD5 and TSS was based on the pragmatic approach developed by USEPA staff to give a definition to the legislative requirement of secondary treatment. The staff developed a statistical basis for determination of the likelihood of meeting various requirements, and the staff conclusion was that most secondary plants could meet the numerical requirements. Subsequently, modifications to the secondary requirements were made when it was recognized that conventional trickling filter plants and oxidation ponds were underrepresented in the original assessment relative to their numbers in actual service, and the possibility for case-specific modifications were made to avoid unnecessary and costly upgrades of these types of plants that otherwise could not meet the “30/30” requirements and where there were no water quality concerns occasioned with the use of higher concentrations (for example, see USEPA, 1984).

Some of the early biological nutrient removal plants, faced with moderate treatment requirements (e.g., monthly values of 10 mg/L TN or 1 mg/L TP) were regulated on a monthly and weekly basis and others on a 12 month rolling average or annual basis. With water quality-based permits requiring some dischargers to reach very low nutrient concentrations, the industry engaged in an effort to define the “Limit of Technology (LOT).” As the comprehensive WERF investigation showed (Bott and Parker, 2011), there is no precise definition of LOT and, in fact, the investigators discouraged the use of this term because it was highly subjective. Instead, WERF used the statistical based approach summarized in Section 3.1 to determine the attainability of various effluent qualities on more precise statistical bases. This reflected the fact that the technologies employed were adapted to meet the regulatory framework (e.g. TMDL or permit limits) and recognized that the variability of each plant was based on the selected technology, service area and waste and flow loading characteristics. These empirical findings were of great significance to engineers and regulators for that reason.

Tables 3-3 and 3-4 summarize the plants and records investigated by WERF. The plants are characterized in terms of how the nutrients are removed. For nitrogen removal plants, there are three types:

- A combined nitrogen removal plant removes all the nitrogen in a single activated sludge process.
- A separate stage nitrogen removal process has two distinct processes in separate steps, one for nitrification and one for denitrification.
- A multiple stage nitrogen removal process utilizes several processes to remove nitrogen (e.g. a combined process followed by a separate stage process.

The phosphorus removal plants are also categorized into two types, chemical addition and biological phosphorus removal. Chemical removal plants were subdivided into a single stage plant that adds chemicals at one point, whereas a multiple stage plant uses at least two chemical addition points. Although the single and multiple stage plants may also rely in part on biological phosphorus removal to reduce chemical requirements, the third type of plant examined by WERF was a biological phosphorus removal plant that normally does not rely on chemical addition at all.

Table 3-5 summarizes the permit limitations that were operative during the period of plant operations analyzed, for all but the single Canadian plant evaluated by Bott and Parker (2011). This table demonstrates that, for plants regulated for both TN and TP simultaneously, not a single plant had very low limits in both categories. Therefore, those plants were carried on one list or the other (Table 3-3 or 3-4) and the data for the nutrient with the lowest limit were analyzed. Because there were no plants with very low limitations for both TN and TP, the treatment requirements for obtaining low values for both can only be inferred and cannot be based directly on the WERF results. Existing permits have a diversity of averaging periods regulating nutrients, as shown in Table 3-5. More plants are regulated on an annual basis than on a daily basis, because the impacts of nutrients on eutrophication are more often cumulative rather than sensitive to extreme values, as would be the case to toxics regulation.

Table 3-3. Summary of Nitrogen Removal Plants Investigated (after Bott and Parker, 2011)

Plant	Nitrogen Removal Process Type	Cold or Warm	Primary Treatment	Secondary Treatment	Tertiary Treatment
River Oaks, FL	Separate Stage N removal	Warm	Clarifiers, EQ	Aeration Tanks in Series, Clarifiers	Denitrification Basins, Clarifiers, Dual Media Deep Bed Filters
Eastern WRF, FL	Combined N removal	Warm	None	5-Stage Bardenpho Carrousel, Clarifiers	ABW Filters
Parkway, MD	Combined N removal	Cold	Clarifiers	4-Stage Bardenpho, Clarifiers	None
Fiesta Village, FL	Multistage N removal	Warm	None	Oxidation Ditches, Clarifiers	Denitrification Filters
Western Branch, MD	Separate Stage N removal	Cold	None	HRAS, Clarifiers, NAS, Clarifiers, DNAS, N_2 Stripping Channel, Clarifiers	Dual Media Gravity Filters
Scituate, MA	Separate Stage N removal	Very Cold	None	Aeration Tanks, Clarifiers	Denitrification Filters
Truckee Meadows, NV	Separate Stage N removal	Cold	Clarifiers	Aeration Basins, Clarifiers	Nitrifying Trickling Filters, Denitrifying Fluid Bed Reactors, Dual Media Gravity Filters
Piscataway, MD	Multistage	Cold	Clarifiers	Step Feed Biological Nutrient Removal, Clarifiers	Dual Media Gravity Filters
Tahoe-Truckee, CA	Separate Stage N removal	Very Cold	Clarifiers	HPOAS, Clarifiers	Floc Basins, Chemical Clarifiers, Recarb Basins, Clarifiers, Recard Basins, Ballst Ponds, BAF, Tertiary Filters, Disinfection, SAT

Table 3-4. Summary Phosphorus Removal Plants Investigated (after Bott and Parker, 2011)

Plant	Phosphorus Removal Process Type	Cold or Warm	Primary Treatment	Secondary Treatment	Tertiary Treatment
Iowa Hill, Co	Single Stage Chemical Addition	Very Cold	None	Anaerobic Zones, Aeration Basins, Clarifiers, EQ	Fine Screening, BAFs, DensaDeg Chem P Removal, Continuous Backwash Upflow Sand Filters
F. Wayne Hill, GA[1]	Multiple Stage Chemical Addition	Moderate	Clarifiers	Aeration Basins, Clarifiers, EQ	Chemical Clarifiers, Deep Bed Granular Media Filters
F. Wayne Hill, GA[1]	Multiple Stage Chemical Addition	Moderate			Chemical Clarifiers, Ultrafiltration Membranes
Cauley Creek, GA	Single Stage Chemical Addition	Moderate	None	Modified Johannesburg BNR	Membrane Bioreactor
Clark County, NV[1]	Multiple Stage Chemical Addition	Moderate	Clarifiers	Anaerobic/Oxic Basins, Clarifiers	Dual Media Filters
Clark County, NV[1]	Multiple Stage Chemical Addition	Moderate			Chemical Clarifiers, Dual Media Filters
Rock Creek,OR[1]	Multiple Stage Chemical Addition	Very Cold	Clarifiers	Step Feed MLE Aeration Basin, MLE Aeration Basins, Clarifiers	Upflow Floc Blanket Clarifiers, Monomedia Gravity Filters
Rock Creek,OR[1]	Multiple Stage Chemical Addition	Very Cold		MLE Aeration Basins, Clarifiers	Chemical Clarifiers, Dualmedia Gravity Filters
Blue Plains, DC	Multiple Stage Chemical Addition	Cold	Clarifiers	Activated Sludge, Clarifiers	Nitrification and Denitrification Reactors, Clarifiers, Multimedia Filters
ASA, VA	Multiple Stage Chemical Addition	Cold	Clarifiers	Step Feed Biological Reactor Basins, Clarifiers	Rapid Mix and Flocculation, Inclined Plate Settlers, Gravity Filters
Pinery, CO	Single Stage Chemical Addition	Cold	None	5-Stage Bardenpho Process, Clarifiers, EQ	Trident Adsorption Clarifier-Filter Process
Kelowna, BC	Little or No Chemical Addition	Cold	Clarifiers, EQ	3-Stage Bardenpho Process, Clarifiers	Dual Granular Media Gravity Filters
Kalispell, MT	Little or No Chemical Addition	Very Cold	Clarifiers, EQ	Modified UCT Process, Clarifiers	Gravity Sand Filters

Note: [1]These plants have two separate types of treatment trains.

Table 3-5. Summary of Nutrient Permit Requirements for Operating Period of Plants Investigated (after Bott and Parker, 2011)

Plant	Annual	Seasonal	Month	Week	Max Day
River Oaks, FL					
TN, mg/L	3.0	na	3.75	4.5	6.0
TP, mg/L	1.0	na	1.25	1.5	2.0
Eastern WRF, FL					
TN, mg/L	3.0	na	5.0	6.0	na
TP, mg/L	1.0	na	2.0	2.4	na
Parkway, MD					
TN, mg/L	na	4/1 to 10/15 for month and week	7.0	11.0	na
TP, mg/L	na	na	1.0	1.5	na
Fiesta Village, FL					
TN, mg/L	3.0	na	3.0	4.5	6.0
TP, mg/L	0.5	na	0.5	0.75	1.0
Western Branch, MD					
TN, mg/L (kg/d)	na	4/1 to 10/31 for month and week	3.0 (340)	4.5 (510)	na
TP, mg/L (kg/d)	na	na	1.0 (110)	na	na
Scituate, MA					
TN, (lb/d)	(53a)	na	(53a)	na	na
Truckee Meadows, NV					
TN, (lb/d)	(500)	Monthly: May to October	(500)	na	na
TP, (lb/d)	na	na	(134)	na	na
Piscataway, MD					
TN mg/L (lb/yr)	(513,800)	na	na	na	na
TP mg/L (lb/d)	na	na	0.18 (45)	na	na
Tahoe-Truckee, CA					
TN, mg/L	3.0	Monthly: May 1 to October 31	2.0	na	na
TP, mg/L	0.3	na	na	na	na
Iowa Hill, CO					
TP, mg/L (lb/yr)	(225)	na	na	na	0.5
F. Wayne Hill, GA					
TP, mg/L	na	na	0.13	na	na
Cauley Creek, GA					

TP, mg/L	na	na	0.13	na	na
Clark County, NV					
TP, mg/L (lb/day)	na	na	176	na	(176)b
Rock Creek, OR					
TP, mg/L	na	Monthly: May 1 to October 31	0.10c	na	na
Blue Plains, DC					
TP, mg/L	0.18a	na	na	0.35	na
ASA, VA					
TN, mg/L	6.0	na	na	na	na
TP, mg/L	na	na	0.18	0.27	na
Pinery, CO					
TP, mg/L (lb/day)	(304)	na	0.05	na	0.1
Kalispell, MT					
TP, mg/L	na	na	1.0	na	na

Note: a. 12 month rolling average, b. Waste load allocation, c. Monthly median

Because the majority of plants surveyed were permitted on the basis of monthly permits, it was deemed by the WERF investigators that all the technologies should be compared on the basis of the 95 percentile of the monthly average data (Table 3-6). This percentile represents exceedance of the monthly average limit slightly less than once annually (actually three times per permit period). For nitrogen removal plants, WERF also concluded that examination of an earlier Florida survey by Jimenez et al., 2007 allowed a better characterization of performance differences. Overall, the separate and multiple stage processes outperformed the combined processes, likely because of the ability to tailor the addition of chemicals (e.g. methanol) more precisely than is possible when all the reactions take place in a single activated sludge process.

The earlier USEPA survey (Kang and others, 2008) found 95^{th} percentile monthly values for nitrogen in cold climates of no lower than 4.2 to 4.9 TN mg/L. However, the WERF investigators found Kalkaska's excellent performance on TIN for a combined plant in a cold climate condition, reaching the preliminary conclusion that it could usually meet an average monthly limit value of 3.2 mg/L TN, pending the collection of additional data. Because the combined nitrogen removal plants are often less reliant on organic chemical addition to achieve low levels of nitrogen in their effluents, it emphasizes the point that just 1-2 mg/L variation in the monthly average effluent requirement can make a large difference in the sustainability of the wastewater treatment process in terms of minimizing the use of chemicals.

Table 3-6. 95th Percentile Monthly Average TN for Three Categories of Nitrogen Removal Plants (after Bott and Parker, 2011)					
Separate Stage	TN, mg/L	Combined	TN, mg/L	Multiple Stage	TN, mg/L
River Oaks, FL	2.3	Kalkaska, MI [a]	1.7	Fiesta Village, FL (Denite Filter)	2.2
Western Branch, MD	2.4	Parkway, MD	5.1	5 A2/O Plants with Denite Filters, FL[b]	3.0
Truckee Meadows, NV	2.5	Eastern WRF, FL	6.7		
Tahoe-Truckee, CA	3.1	Piscataway, MD	7.2		
Scituate, MA	3.8	10 Bardenpho Plants, FL[b]	3.5		
Howard F Curran, FL[b]	3.0				

Note: a. Kalkaska has a TIN based permit; assuming ON value of 1.0 to 1.5 mg/L, the TN value could be 2.7 to 3.2 mg/L. It was evaluated as a nitrification plant in the WERF report and was not included in Table 3-5. b. Data for these plants are not included in the WERF report and are from Jimenez et al., 2007.

For the phosphorus removal plants evaluated, the single stage chemical addition plants outperformed the other classes of plants, but at the expense of much higher chemical dosages (see Bott and Parker, 2011 for chemical types and requirements). It was theorized that the multiple stage plants would have performed closer to the single stage plants if their actual effluent requirements had been closer. The plants relying exclusively on biological phosphorus removal, while performing quite well, could not reach as reliably low TP levels as the other two categories of plants. However, when comparing the plants on an annual performance, basis, the Kalispell plant performs as well as some of the plants in the other two categories (see Bott and Parker, 2011).. This is an important finding, because if regulation can be on an annual basis for phosphorus and still properly address eutrophication concerns, a more sustainable treatment technology might be provided. That is there is a greater chance that chemical addition can be avoided and reliance can exclusively be made on biological means for phosphorus removal.

Table 3-7. 95th Percentile Monthly Average TP for Three Categories of Phosphorus Removal Plants (after Bott and Parker, 2011)					
Multiple Stage	TP, mg/L	Single Stage Chemical Addition	TP, mg/L	Little or No Chemical Addition	TP, mg/L
F. Wayne Hill, GA	0.0902	Iowa Hill WRF, CO	0.0306	Kalispell, MT	0.168
ASA, VA	0.101	Pinery, CO	0.0363	Kelowna, BC	0.217
Clark County, NV	0.153	Cauley Creek, GA	0.116		
Rock Creek, OR	0.151				
Blue Plains, DC	0.161				

Bott and Parker, 2011 concluded: “A major finding of the WEF/WERF investigation was that statistical variability is a characteristic of all the exemplary plants and that this variability should be recognized in both evaluation of technologies (e.g., stratifying them in terms of their capabilities) in an engineering environment as well as determining the appropriate effluent limits in the regulatory permit setting environment.”

Further, they found: “It is the obligation of the regulators, regulated community, and the design engineering profession to recognize the process variability and higher risks that are attendant with the design for very low nitrogen and phosphorus concentrations or very low maximum day ammonia concentrations. When designing for typical secondary treatment requirements, high effluent concentration days can be balanced against low effluent concentration days. When designing for concentrations close to zero, it would require negative concentrations (which do not exist) to provide similar risk mitigation as occurred in the past with conventional secondary treatment. The research shows that when seeking very low effluent nutrient levels, even exemplary plants may produce violations if regulators pick values that are inappropriate to the associated averaging period, no matter how much effort, cost and energy is expended. The goal for regulators, operator and plant designers should be to assure the public that the investment of public dollars can properly be done by finding statistical bases for regulation that are both protective of the environment and are technologically achievable.”

Although the WERF work was not oriented to evaluation of the appropriateness of the TSD approach for nutrients, these conclusions are highly relevant to the derivation of both WQBELs and technology-based limits for nutrients.

Section 4

Review of TSD Permitting Components

The heart of this technical review is a systematic evaluation of the TSD's recommended permitting approaches for toxics, in light of their potential application to nutrients. As discussed in section 1.4, the basic review approach was to deconstruct the TSD approach into individual components, and to classify those components as appropriate, appropriate in certain circumstances, appropriate with recommended modifications, or inappropriate for nutrients. Reviewers also considered elements that are necessary for nutrients but are not fully addressed in the TSD or otherwise typically considered for toxics. For organizational purposes, the elements of the TSD were grouped into three major categories: (1) determining when a limit is needed; (2) deriving WLAs; and (3) deriving WQBELs.

4.1 Determining When a Limit is Needed

The TSD cites federal regulations [40 CFR 122.44(d)(1)] as requiring permitting authorities to evaluate whether a discharger has the "reasonable potential" to cause or contribute to an excursion of numeric or narrative water quality criteria. If reasonable potential is found, the permitting authority must include a limit in the NPDES permit for that parameter. As discussed in section 1.1.2, this language was primarily developed with toxic constituents in mind. Regardless, permitting agencies will face the decision of when to include WQBELs for nutrients in NPDES permits, or when no limits are needed to protect water quality standards.

The TSD document and USEPA training materials present two basic approaches for determining reasonable potential, one of which is performed in absence of effluent monitoring data (qualitative approach); and the other of which is performed using effluent monitoring data (quantitative approach). Both of these approaches were evaluated for appropriateness with nutrients.

4.1.1 Qualitative or Semi-Quantitative Methods

In the absence of facility-specific monitoring data, the TSD outlines a number of considerations that the permitting authority might use to determine reasonable potential, such as:

- The amount of dilution expected in the receiving water body.
- The type of industry or POTW.
- Existing data on other toxic pollutants.
- History of compliance problems and toxic impacts.
- Type of receiving water body and designated use, including current impairment or priority status.

Under this procedure, the reasonable potential analysis (RPA) is a professional judgment based on weight-of-evidence. The presence of multiple factors—such as low dilution, a receiving water that is already impaired, and/or poor compliance record—would increase the likelihood that the discharge would be deemed to have reasonable potential to cause or contribute to an excursion.

This approach is primarily intended as a substitute for the ability to quantitatively predict the toxic concentration in the receiving water, and so this approach is sometimes labeled as a “qualitative” reasonable potential analysis (RPA). Some of the considerations may have quantitative elements (e.g., dilution, existing water quality), and so it can also be characterized as a “semi-quantitative” approach. The TSD states that “a clear and logical rationale for the need for limits” is necessary.

For nutrients, permitting agencies may also encounter situations where the concentrations of nutrients or response variables in the receiving water cannot be predicted in a highly quantitative fashion. With toxics, the issue for most discharges is a lack of information regarding the presence or absence of, or the actual concentrations of, a particular parameter. This is not likely to be the case for most discharges in regards to nutrients. Rather, permitting agencies are more likely to consider qualitative or semi-quantitative RPAs where:

- Nutrient criteria are narrative rather than numeric; or
- No calibrated model(s) exist to quantify relations between nutrient inputs and receiving water responses.

In general, the complexities of eutrophic responses are such that purely qualitative RPAs are not recommended for nutrients. The most defensible RPAs will be quantitative, based on calibrated load-response models, as discussed in the following section. If semi-quantitative RPA approaches are used, they must be objective, reproducible, considerate of the assimilative capacity, and take into account the major factors that controls eutrophic responses of the receiving water.

It is recommended that if permitting agencies pursue semi-quantitative RPA approaches for nutrients, they do so using documented procedures that are subject to public review and comment. Although it is beyond the scope of this review to develop such procedures for multiple water body types, it is recommended that they include elements such as the following:

- Clear mechanisms for interpreting narrative nutrient criteria that include response variable endpoints.
- Consideration of the current status of the receiving water body with respect to response endpoints, including biological status. If the receiving water is healthy or otherwise meeting designated uses, the reasonable potential of impairment from existing dischargers is low.
- Semi-quantitative evaluation of other receiving water body characteristics that could affect responses to nutrients and assimilative capacity, including light availability, hydraulic characteristics, temperature.
- Historic trends in water quality and response variables.
- The magnitude of the proposed discharge relative to receiving water body flow under critical conditions appropriate for nutrients.
- The magnitude of the proposed nutrient load from point sources relative to loads from other sources including background.
- Existence of a nutrient TMDL or similar basin plan that controls nutrients.

The state of Texas provides an example of documented, semi-quantitative screening procedures for nutrient RPAs. The *Procedures to Implement the Texas Surface Water Quality Standards* (TCEQ, 2010) outlines a screening approach for determining whether nutrient limits are necessary.

The procedure includes relatively simple computations to ensure that potential increases in nutrients and chlorophyll-*a* in reservoir main pools would be relatively small. Although the equations do not represent calibrated models, they allow conservative estimates of phosphorus losses in tributary streams, phosphorus delivery rates to reservoirs, and potential changes in chlorophyll-*a*. The Texas procedure also allows the use of alternative models/evaluations if they are available.

Another example of a potential semi-quantitative RPA approach was provided as a case study in the recent WERF report on modeling site specific nutrient goals (Bierman and others, 2013). The case study—entitled *Screening-Level Modeling of Stream Algal Benthic Responses, Virginia*—demonstrated the concept of using screening-level models to evaluate whether receiving streams would or would not be sensitive to changes in nutrient availability. The authors concluded that because benthic algae estimates were sensitive to key model parameters (especially growth rate limitation terms), it would be recommended to first calibrate such models at representative regional locations before using them in a semi-quantitative fashion for regulatory purposes. If this were done, such models could conceivably be used to identify waters that have assimilative capacity, and those that would be more likely to be adversely affected by new or expanded nutrient discharges.

Element of Toxics-Based Permitting	Qualitative or semi-quantitative methods for determining whether a NPDES limit is needed.
Summary of USEPA Training Slide Content	Discusses options for performing qualitative RPA, many of could actually be characterized as semi-quantitative because they utilize effluent and water quality data.
Conclusion of Review	**Appropriate with modification** for nutrients. Quantitative RPA is preferred, and purely qualitative RPA methods are discouraged for nutrients. However, semi-quantitative RPA can represent a reasonable approach where calibrated models are not available.
Recommended Modifications of TSD Approach	Semi-quantitative RPA approaches for nutrients need to be objective, reproducible, consider assimilative capacity, take into account all factors, and be subject to public review/comment. Various characteristics of the receiving water should be considered to evaluate assimilative capacity, including current impairment status. Screening-level modeling approaches have potential to provide valuable information, lacking calibrated models. A purely qualitative approach is not recommended.

4.1.2 Quantitative Methods

The TSD's recommended approach for quantitative RPA is statistical in nature, and involves comparison of an estimate of the upper bound of the expected concentration in the receiving water with numeric water quality targets. The TSD illustrates a method for estimating the upper bound as a 95th or 99th percentile of the effluent data at a 95th or 99th level of confidence. If sufficient data ($n > 10$) are available, it is recommended to calculate a Coefficient of Variation (CV) from the effluent data. If insufficient data are available, the default assumptions include a lognormal distribution of effluent data and a coefficient of variation (CV) of 0.6. The TSD includes tables with multiplication factors to estimate the upper bound of the effluent data from the highest value in the observed data set. These values would then be used with dilution factors to estimate the maximum receiving water concentration. USEPA recommends a finding of reasonable potential when the maximum receiving water concentration exceeds the water quality criterion.

There are several reasons why this toxics-based RPA method is not appropriate for nutrients.

Nature of Nutrient Impacts: As described in the TSD, the quantitative RPA approach does not directly consider the averaging period of water quality criteria. The resulting estimates of the upper bound concentrations will be higher than if averaging periods were applied. This is not problematic for constituents with very short averaging periods (e.g., an acute with a 1-hour averaging period) because the upper bound is still a reasonable (if conservative) estimate of the maximum concentration that could be experienced within that short averaging period.

However, the longer the appropriate averaging period, the more the upper bound calculated in this manner represents an unreasonably conservative estimate of the maximum receiving water concentration at the appropriate averaging period.

As discussed in Section 2, nutrient impacts in hydrologic systems are the result of loading over longer time periods (weeks to months, seasonal, or annual) rather than the one-hour to 4-day averaging periods that are relevant to toxic impacts. The occurrence of short-lived “spikes” of upper-bound concentrations is much less relevant for nutrient permitting than for toxic permitting. Rather, the result of interest for the RPA is the maximum concentration or load over the appropriate averaging period. This assumes the availability of valid concentration targets (linked to designated uses) that have magnitude, frequency, and duration components. The need to modify the TSD approach is reflected in slide 86 of the USEPA training materials, which recommends the use of “different upper bound effluent pollutant concentrations to use with different criteria durations”. If valid nutrient concentration targets are available, it is recommended that a quantitative RPA employ an upper-bound estimate of the appropriate time-averaged concentration in the receiving water. Some states already use a version of this approach by using time-averaged data (e.g., monthly averages) from discharge monitoring reports (DMRs) to perform the reasonable potential analysis.

Preference for load-response linkages: As discussed in section 4.2, the use of numeric nutrient criteria and load-response linkages represent two different approaches for controlling excessive nutrient loadings. Load-response modeling represents a viable alternative to the TSD approach for quantitative RPA, particularly for settings where the receiving water responses of interest are complex functions of multiple sources or environmental factors. Load-response modeling—or even screening-level modeling methods described in section 4.1.1—would be preferred over the use of questionable concentration targets (e.g., USEPA’s 2000 ecoregional criteria or 1986 Gold Book values) with the TSD’s standard quantitative TSD approach.

Calibrated load-response models can be used to directly determine when a proposed discharge or combination of discharges has the reasonable potential to cause undesired responses such as excessive algal growth, low dissolved oxygen, poor water clarity, etc. Models can also be used to quantify how much of the available assimilative capacity that a proposed discharge would use. If using load-response linkages for RPA, it is recommended that there be documentation of an objective means for their use, such as quantitative definitions of what kinds of predicted responses would indicate reasonable potential.

Element of Toxics-Based Permitting	Quantitative method for determining whether a NPDES limit is needed; specifically, the comparison of upper-bound estimates to water quality criteria.
Summary of USEPA Training Slide Content	Describes a method very similar to the approach recommended for toxics, using nutrient concentration targets. Discusses options for selecting default concentration targets where state NNC are lacking, such as the 1986 Gold Book or USEPA ecoregional criteria. Mentions alternative to use an "indicator parameter". Discusses modeling options in broad terms.
Conclusion of Review:	**Inappropriate for nutrients** in situations where valid concentration targets (linked to designed uses) are lacking. **Appropriate with modification** for nutrients if valid concentration targets (linked to designated uses) are available.
Recommended Modifications of TSD Approach	If valid concentration targets are available, TSD methods need to be modified to estimate the upper bound of the effluent concentration using the appropriate averaging period. Time-averaged concentrations will usually be significantly lower than the upper bounds estimated from the unmodified TSD approach. If valid concentrations are not available, it is recommended to use calibrated load-response models or semi-quantitative screening approaches for RPA.

4.2 Deriving Wasteload Allocations

A wasteload allocation (WLA) is the portion of a receiving water's assimilative capacity that is assigned to a point source. Methods for deriving WLAs for toxics can vary from simple mixing calculations to complex models. The derivation of science-based, equitable WLAs is arguably the single most important step in the permitting process. Appropriate WLA derivation methods for nutrients may differ from those for toxics in several different arenas, including:

- Use of concentration targets versus load-response linkages
- Modeling approaches
- Appropriate critical conditions
- Averaging periods
- Consideration of mixing/dilution
- Nutrient-specific considerations such as nutrient ratios and bioavailability.

This subsection evaluates the manner in which WLA derivation methods for nutrients be different or similar to those for toxic constituents.

4.2.1 Deriving WLAs from Nutrient Concentration Targets versus Load-Response Linkages

One of the most fundamental assumptions of TSD-based permitting methods is that of dose-response. The permitting methods described in the TSD largely presume the availability of in-stream water quality criteria expressed as concentrations. As discussed in Section 2, concentration-based criteria for toxics have relatively short averaging periods, such as a 1-hour average for acute criteria and a 4-day average for most chronic criteria. WLAs for toxics normally derived to prevent excursion of acute and chronic concentration targets in the receiving water body.

In contrast with toxics, the conceptual models of nutrient impairment are more complex than simple dose-response. Although some individual nutrient species (e.g., ammonia, nitrate) can cause direct toxic effects, most nutrient management efforts are aimed at limiting the value of intermediate response variables such as algal biomass or water clarity, which can impair designated uses in a number of manners as discussed in Section 2.

Nutrient concentrations can provide a measure of short-term algal growth potential. However, the nature of system-specific responses can depend on a great many factors other than solely nutrient concentrations (e.g., hydraulic residence time, light availability, temperature). Although nutrient concentration targets may be valid in some settings, other systems will be more appropriately managed by direct linkage of nutrient loading to response variable endpoints.

4.2.1.1 Situations Where WLAs Can Be Based On Valid Nutrient Concentration Targets

Concentration-based water quality targets may be more appropriate for some systems where there is a demonstrable and consistent relation between nutrient concentration and the response of interest. Often this relation will be water-body specific. Examples of systems where concentration-based goals are appropriate include:

- Lakes and reservoirs that have a definable relation between seasonal average phosphorus concentrations and algal biomass, as determined from empirical data and/or models. Such relations have been demonstrated in various geographies (e.g., Dillon and Rigler, 1973; Trowbridge, 2009), and can vary a great deal between water bodies even with the same geography (Hoyer and Jones, 2011; Brett and Benjamin, 2008).
- Wadeable stream systems where algal biomass accrual potential is a function of nutrient concentration under specific conditions, such as stable flow, or high-temperature periods. Such relations have been demonstrated empirically (e.g., Biggs, 2000; Dodds and Welch, 2000) or derived from models (Flynn and Suplee, 2013). As in lake/reservoir settings, such relations are subject to water body-specific variation, and not all streams have useful empirical relations between nutrients and algal biomass (Morgan and others, 2006).

For the purposes of this review, it was recognized that the availability of valid nutrient concentration targets could vary a great deal between regulatory settings. It is presumed that there are certain situations where valid nutrient concentration targets can be identified and are appropriate for use in permitting. In this review document, the term “valid” is used to denote concentration targets that:

- Are based on cause-and-effect, with a demonstrated relation between the concentrations and response of interest; and
- Have magnitude, frequency, and duration components, all based on the ecological responses of interest.

The validity of NNC and associated derivation methods has been one of most difficult and controversial aspects of USEPA’s National Nutrient Strategy. USEPA’s 2000 ecoregional criteria—or state criteria derived using similar methods—are not presumed to represent “valid” concentration targets as defined above. Among other reasons, the percentile-based method used in their derivation does not directly address cause-and-effect and can potentially result in highly overprotective criteria. Similarly, the Gold Book (USEPA, 1986) values discussed in the USEPA training materials lack frequency/duration components and cannot account for the wide variety of characteristics of different receiving water bodies. The Gold Book values may be over- or under-protective depending on the system.

More rigorous means for deriving valid concentration targets include stressor-response methods (USEPA, 2010b) and mechanistic modeling. In their review of the stressor-response guidance, the USEPA Science Advisory Board (2010c) has correctly cautioned that the statistical associations do not necessarily represent cause-and-effect, and recommended that stressor-response analysis be combined with other available information in a weight-of-evidence approach. If sufficient resources are available, calibrated models and data analysis provides a means to identify concentrations targets with frequency and duration components.

WERF has recently provided guidance on how models may be used to derive water body-specific nutrient goals, including numeric nutrient criteria (Bierman and others, 2013). This document recommends the following steps to derive the NNC from calibrated models:

- Verify the concentration-response relation using both model output and observed data.
- Resolve the spatial and temporal components of the model scenarios/output with the intended regulatory assessment approach.
- Test candidate NNC for Type I and Type II errors; *i.e.*, false findings of attainment or non-attainment of response endpoints.
- Document the technical basis for the NNC, including any caveats and limitations derived from the model.

The outcomes of these steps may lead a project team to conclude that NNC are not the preferred regulatory tool for nutrient management. For example, it may be determined that nutrient concentrations are not reliable predictors of response variables, or that assessment error rates would be unacceptably high. In this situation, the project team might conclude that the system is best managed by loads, adaptive management methods, or bioconfirmation methods.

Using existing ambient concentrations to develop WLAs and WQBELs: Because nutrient concentrations are often a poor predictor of biological impairments, some states (e.g., ME, FL, OH) have developed bioconfirmation methods that utilize a combination of nutrient concentrations and response variables for assessment. If default nutrient concentration targets are exceeded but response variables are favorable, these methods include the option for developing site-specific concentration targets or criteria. For example, under Florida's proposed approach (Florida DEP, 2013), if a stream exceeds default NNC but has healthy biology, WQBELs would be based on existing conditions. In practice, this would mean that existing permit levels would be maintained. New or expanded dischargers would require a more detailed study of appropriate nutrient levels. This use of existing conditions for developing WLAs does not apply to toxics and thus has no parallel in the TSD, but is encouraged for use with nutrients.

4.2.1.2 Situations Where WLAs Need to Be Based On Load-Response Linkages

In many systems, nutrient concentrations will be a highly unreliable predictor of designated use attainment, and nutrients will best be managed by linking nutrient loads to response variables such as dissolved oxygen, water clarity, or chlorophyll-*a*. For example, with regard to predicting eutrophic responses in estuaries and coastal areas, USEPA (2001) states that "nutrient concentration often does not provide a useful relationship". Quoting Dr. Han Paerl, USEPA (2001) states that "whether or not to use concentrations or loading as criteria largely depends on the spatial and temporal scales of assessing ecosystem responses to nutrient inputs". As discussed further in section 4.2.3, water bodies with longer retention times will tend to respond to cumulative nutrient loads over a seasonal or annual basis. Therefore, load-response linkages would be more accurate than concentration targets for most lake, reservoir, and estuarine settings. This is especially true for systems with longer residence times (>90 days) which store nutrients in the water column and sediments, and thus respond to the integrated effect of nutrient loading over seasonal or annual time periods, rather than short-term concentrations.

However, even in short-retention stream systems, nutrient concentration can be a poor predictor of responses. For example:

- Florida DEP (2009) found very weak statistical relations between in-stream nutrient concentration and a periphyton index, and concluded that site-specific variability prevented the development of numeric nutrient criteria on this basis.

- In guidance on developing nutrient TMDLs, USEPA (1999) concluded that "phosphorus indicators are not as easy to implement in rivers and streams as they are in lakes and reservoirs…The relationship between phosphorus concentration and plant growth is not as well established…"
- Broadly summarizing the experience of many jurisdictions, the Association of State and Interstate Water Pollution Control Administrators (ASIWPCA) (now the Association of Clean Water Administrators) concluded that "During there considerable development processes, many States are failing to find a strong linkage between the EPA recommended causal variables (N and P) and response variables…In many cases, such as relationship cannot be demonstrated…These problems can only lead to miscues in impairment identification and misdirection of scare management and implementation resources." (ASIWPCA, 2007).
- Studying mountain/foothill streams and rivers in Colorado, Lewis and McCutchan (2010) found "no meaningful relationship between periphyton biomass accumulation and concentrations of total or dissolved forms of nitrogen or phosphorus."
- Macrophytes can obtain nutrients from the sediment, and therefore are not highly responsive to water column concentrations (Hilton and other, 2006; Wisconsin DNR, 2012).
- California's numeric nutrient endpoints framework is based on the premise that "biological response indicators are better suited to evaluate the risk of beneficial use impairment, rather than using pre-defined nutrient limits that may or may not result in mitigation of eutrophication for a particular water body." (Sutula and others, 2007).

In guidance on developing WLAs for lakes and reservoirs, USEPA (1983) emphasizes the development of mass loading estimates to control response variables, stating:

> "Loading estimates for nutrient inputs to lakes are required for all of the analysis frameworks available to examine waste load allocations…the loading estimates should define *mass inputs* [emphasis added] of the limiting nutrient by type and location of a source."

The USEPA's Science Advisory Board (USEPA SAB, 2010c) concluded that numeric nutrient concentration criteria and load-response linkages should be considered as independent approaches for controlling excessive nutrient loadings. In making this statement, the SAB did not consider the derivation of concentration-based targets to be a necessary step of nutrient control measures, if load-response linkages are available. This concept is supported by many examples of successful TMDLs and related basin planning efforts that have derived WLAs from load-response linkages rather than ambient nutrient concentration targets.

Some examples include:

- The Chesapeake Bay TMDL (USEPA, 2010a) expressed wasteload allocations (WLAs) as average annual mass loading rates to achieve criteria for dissolved oxygen, water clarity, and chlorophyll-*a* concentrations. Ambient nutrient concentration targets were not a component of the Chesapeake Bay TMDL.
- The approved TMDL for Green Lane Reservoir in Pennsylvania (Tetra Tech, 2003) is representative of many lake/reservoir TMDLs in that that its goal was to "reduce phosphorus *loadings* [emphasis added] to the lake so that chlorophyll-a levels [the reservoir] stay at or below [the target chlorophyll a level] as a seasonal average." No phosphorus concentration targets were used.

- Colorado has control regulations for several reservoirs which allocate an annual loading for phosphorus to dischargers in each reservoir's watershed. The goal of these Control Regulations is to limit the amount of algae growth to protect the water supply, recreational and aquatic life designated uses. These Control Regulations do also cap the concentrations that can be discharged, and both the concentration and allocated loading values are included in permits for the associated facilities.
- North Carolina has established nutrient control programs for multiple water bodies (Neuse River, Tar-Pamlico River, Jordan Lake, etc.) based on the linkage between nutrient loads and chlorophyll-a targets, without the use of nutrient concentration targets.

Load targets can also have several practical benefits over ambient concentration targets for permitting. Loading goals can be easier to calculate from models, and are inherently more compatible with water quality trading/offset programs.

Using existing TMDLs to develop WLAs and WQBELs: Under Florida's proposed approach implementing NNC (Florida DEP, 2013), the WLA from an existing TMDL or established watershed plan can take precedence over default NNC for nutrient permitting. Similarly, an existing reasonable assurance (RA) plan can serve as the numeric interpretation that the nutrient standards will be met. This approach acknowledges that watershed-specific evaluations and load-response linkages are superior for managing nutrients than default concentration-based targets, and has no direct parallel in the TSD.

Element of Toxics-Based Permitting	Assumption of dose-response and availability of valid concentration targets (linked to designated uses).
Summary of USEPA Training Slide Content	Assumes that nutrient concentration targets are available or can be selected by permit writer. Discusses options for selecting default concentration targets where state NNC are lacking, such as the 1986 Gold Book or USEPA ecoregional criteria. Mentions alternatives to use an "indicator parameter" or site-specific criteria. Also mentions the option of basing limits on "stressor-response relationships relating downstream response variables or narrative criteria to upstream nitrogen and phosphorus concentrations or loadings."
Conclusion of Review	**Appropriate with modification** for some systems, and **inappropriate** for others. One of the high-level decisions in any permitting application is the determination of whether valid concentration targets are available, or whether the system is best managed using load-response linkages. Load-based goals will generally be more useful for permitting in large, complex systems such as estuaries, coastal areas, and linked river-reservoir systems, and more conducive to support water quality trading programs.
Recommended Modifications of TSD Approach	If valid concentration targets are available, frequency/duration components need to be selected appropriately. If system responds primarily to loads, WLAs need to be based on load-response linkage instead of concentration targets.

4.2.2 Wasteload Allocation Modeling Methods

The TSD document does not go into great detail regarding the complexities of water quality modeling. However, many permitting authorities normally rely on relatively simple methods for predicting the concentrations of toxic constituents in receiving waters. Often these are simple mixing calculations that calculate the receiving water concentration as function of a background flow/concentration and the effluent contribution. This can be appropriate for conservative toxic constituents that are permitted based on local effects where other loss/transformation terms are not a major factor. However, such simple methods would not be appropriate for nutrient WLAs based on far-field, regional, or downstream impacts.

Nutrients are less likely than most toxics to behave in a conservative fashion in the receiving water, and can be strongly affected by biological uptake and settling even over modest distances. For this reason, it is recommended that nutrient WLA models be complex enough to consider the non-conservative behavior of nutrients in the receiving water. As stated the Permit Writer's Manual (USEPA, 2010d),

> For pollutants such as BOD, nutrients, or non-conservative parameters, the effects of biological activity and reaction chemistry should be modeled, in addition to the effects of dilution, to assess possible impacts on the receiving water.

The TSD approach is generally based on the assumption that WLAs are derived to prevent excursions of in-stream concentration targets for the toxic constituent being permitted. As discussed in section 4.2.1.2, it is often more useful to base WLAs on load-response linkages rather than in-stream nutrient concentration targets. This is especially the case for systems that respond more to longer-term nutrient loads than short-term nutrient concentrations, or for which nutrient concentration is not a highly useful predictor of designated use impairments. For obvious reasons, if a load-response WLA modeling approach is taken, it must be capable of predicting the response variables of interest with reasonable accuracy. Both USEPA (1997) and WERF (Bierman and others, 2013) have provided guidance on the use of load-response models to determine nutrient goals.

As discussed in the TSD, the use of steady-state models versus dynamic models represents a fundamental dichotomy in WLA derivation methods. Steady-state models set constant or time-averaged inputs for effluent flow, effluent concentration, receiving water flow, and other conditions such as temperature and pH. Dynamic models, in contrast, consider the actually variability in receiving water flow and other environmental conditions, and can calculate a frequency distribution of either nutrient concentrations or response variables. Based on the results of this review, either steady-state or dynamic WLA techniques can be used for nutrients, depending on the hydrologic and regulatory setting. However, there are specific circumstances when steady-state WLA methods would be inappropriate, and dynamic WLA techniques would be preferred. Both steady-state and dynamic WLA techniques for nutrients are discussed in subsections below.

4.2.2.1 Steady State Models

Steady-state WLA models can be appropriate for nutrients in simpler settings where a reasonable critical condition can be defined in steady state terms, such as stable growing-season streamflow or a seasonal (quasi-steady state) average condition. However, steady state models can be overly simplistic for nutrients in complex hydrologic settings. Complex systems such as linked river-reservoir systems and estuaries are inherently dynamic, can experience large interannual variations in multiple environmental forcing factors, and may not lend themselves to steady-state analyses. For such systems, dynamic WLA models would be preferred.

Steady state models are inherently highly conservative because WLAs are based on the combined assumptions of maximum effluent flow/load under critical receiving water conditions, such as 7Q10 streamflow for chronically toxic constituents. The use of steady-state model techniques is akin to basing the WLA on the "single, worst-case condition". The TSD acknowledges the following regarding steady state analyses:

> Each condition by itself has a low probability of occurring; the combination of conditions may rarely or may never occur… permit limits [derived from steady-state WLA models] may be more stringent than necessary…In general, steady-state analyses tend to be more conservative than dynamic models because they rely on worst case assumptions. Thus, permit limits derived from these outputs will generally be lower than derived from dynamic models.

The potential for steady state models to produce overly conservative WLAs applies to both toxics and nutrients. However, highly conservative permit limits are less appropriate for constituents that would cause non-lethal, gradational, or aesthetics-based impacts than for those that could have severe or acute effects on aquatic life or human health, as is true for nutrients in many hydrologic settings. For these reasons, it is recommended that the steady state condition is not set to such a rare or “worst-case” condition as to result in unreasonably conservative permit limits. The steady state condition for nutrient WLA models should reflect:

- Longer averaging periods than used for toxic constituents.
- Less rare conditions than used for toxic constituents.

Section 4.2.4 provides additional recommendations on critical conditions for nutrient impacts.

4.2.2.2 Dynamic Models

Dynamic WLA models are preferred for nutrients in most moderately complex to complex hydrologic environments. As stated in the TSD:

> …the least ambiguous and most exact way that a WLA…can be specified [is] by using dynamic models…dynamic models account for the daily variations of and relationships between flow, effluent, and environmental conditions and therefore directly determine the actual probability of that a water quality standards exceedance will occur…dynamic models can be used to develop WLAs that maintain the water quality standards exactly at the return frequency requirements of the standards.

The most common dynamic modeling technique is the application of continuous simulation models to historical hydrologic, water quality, and meteorological datasets. By using actual variations in environmental factors, dynamic models do not need to assume a “single, worst-case” condition for permitting. Rather, model output can be post-processed to compute the duration and frequency components of criteria attainment, and WLAs can be derived that would result in acceptable excursion frequencies. Critical conditions can be defined as seasons, years, or multi-year assessment periods for which the model demonstrates that compliance is more challenging.

Element of Toxics-Based Permitting	Derivation of wasteload allocations from steady-state or dynamic models.
Summary of USEPA Training Slide Content	Discusses modeling options in broad terms. Some examples use steady-state models. No explicit discussion of when a steady-state or dynamic model should be selected.
Conclusion of Review	Model-based WLA methods are **appropriate with modifications** for nutrients but must be tailored to the hydrologic system and eutrophic responses of interest. Allocating WLAs for nutrients also requires **additional considerations** such as equity, technological treatment limitations, and cost-effectiveness (see section 4.2.6).
Recommended Modifications of TSD Approach	• Model to response variables unless valid numeric nutrient concentrations targets are available. • Select models based on hydrologic setting, response variables of interest, data availability, etc. Use models capable of predicting the non-conservative behavior of nutrients. • Utilize longer averaging periods (e.g., seasonal, annual) for WLAs • Use dynamic WLA models instead of steady-state WLA models for moderately-complex to complex eutrophic responses (e.g., complex river, reservoir, or estuarine settings). • Consider equity, technological treatment limitations, and cost-effectiveness in deriving WLAs (see section 4.2.6).

4.2.3 Averaging Period

WLAs will usually be expressed as a mass delivered over a certain time period. Legally, WLAs in TMDLs must be expressed as daily loading rates, but they should also be expressed with longer averaging periods that relate to the expected environmental responses (USEPA, 2007). The appropriate averaging periods of WLAs will be a function of the duration components of water quality criteria and the time period over which the receiving water body is expected to respond to inputs. The appropriate averaging period represents one of the most fundamental differences between nutrients and toxic impacts.

Water quality criteria for protecting aquatic life from toxics are usually expressed as acute (i.e., the criteria maximum concentration or CMC) and/or chronic (i.e., the criteria continuous concentration or CCC) values. The CMC is expressed as a 1-hour average concentration, and is intended to prevent rapid-developing adverse effects such as lethality. The CCC is expressed as a 4-day average concentration, and is intended to prevent longer-term effects such as reduced growth or reproduction. Water quality criteria for human health protection can have various duration and frequency components depending on the endpoints and adverse effects considered. Criteria for carcinogens are derived assuming lifetime exposure (i.e., exposure over 70 years).

In contrast with toxic impacts to aquatic life, nutrient impacts are the result of transport and biological uptake processes that occur on the order of weeks (for low retention time systems) to months or even years (for high retention time systems), rather than hours to days as assumed for toxics. For example, in guidance for developing WLAs for lakes and reservoirs, USEPA (1983) states:

> "The time scale over which mass loading estimates should be developed is determined by the retention time of the lake. Generally, annual loading estimates are required [for lakes and reservoirs]. For small lakes or lakes having short detention times, the annual load may have to be subdivided seasonally."

Lakes and reservoirs have long been managed and modeled by use of seasonally or annually averaged conditions. For example, Vollenweider and Kerekes (1982) demonstrated that the trophic response of lakes and reservoirs could be predicted as a function of average annual phosphorus loading and hydraulic flushing characteristics. Commonly-used lake/reservoir TMDL models such as BATHTUB are inherently seasonally-averaged simulations (Walker, 2006). Water quality criteria for lakes and reservoirs are often expressed as seasonal average conditions. For example, in a review of a completed nutrient TMDL, Tetra Tech (undated) found the "the majority of nutrient TMDLs reviewed were expressed as annual loads".

Similarly, regarding the Chesapeake Bay, USEPA (2004) concluded:

> … that Chesapeake Bay and its tidal tributaries in effect integrate variable point source monthly loads over time, so that as long as a particular annual total load of nitrogen and phosphorus is met, constant or variable intraannual load variation…has no effect on water quality of the main bay.

Rivers are highly variable with regard to retention time, and wadeable streams can be generalized to have relatively short retention times. However, even in short retention time rivers and streams, algae can require weeks to accumulate to undesirable levels. Flynn and Suplee (2013) concluded that attached algae could reach nuisance levels in about 14 days in an enriched river system. This would apply to short-retention systems with highly favorable growing conditions, and could be considered the very low end of a response period for nutrients in any hydrologic system. More generally, Biggs (2000) concluded that eutrophic conditions in streams and rivers were more likely to occur when accrual times exceeded 50 days.

If the WLA is based on a concentration target or numeric nutrient criterion with a longer averaging period, the WLA itself must reflect that averaging period. This is true even for short retention time systems or those that might respond to nutrients on a shorter time frame. The reason is that the combination of the criterion magnitude, duration, and frequency is presumed to protect designated uses. For example, the state of Wisconsin Department of Natural Resources (2012) prepared a justification for using longer (e.g., six-month) averaging period for nutrients, and stated the following:

> Averaging periods should be consistent with the technical analysis and rationale supporting the adopting phosphorus water quality criteria…Wisconsin's approved criteria for wadeable streams and nonwadeable rivers were derived using correlations between growing season median phosphorus concentrations and community biotic indicators…If averaging periods…for permits should reflect methods and data used to derived phosphorus criteria, generally a growing season averaging period is warranted.

USEPA has generally been supportive of the use of longer averaging periods for nutrient WLAs and permitting. The USEPA Permit Writers Manual explicitly states that "…states may adopt seasonal or annual averaging periods for nutrient criteria instead of the 1-hour, 24-hour, or 4-day average durations typical of aquatic life criteria for toxic pollutants." (USEPA, 2010d)

In summary, WLA for nutrients need to be specified with averaging periods that reflect the temporal responses of the system and how criteria/loading goals were derived. In most cases, these will be substantially longer than those for toxics. Monthly, growing-season, or annual averaging periods may be appropriate depending the retention time of the system.

Element of Toxics-Based Permitting	1-hour to 4-day averaging periods of criteria/WLA.
Summary of USEPA Training Slide Content	Acknowledges that longer averaging periods are often appropriate for nutrients. Discusses factors for determining when seasonal or annual averaging periods are applicable.
Conclusion of Review:	**Inappropriate** for nutrients, which affect water bodies over longer time scales.
Recommended Modifications of TSD Approach	Nutrient criteria/WLA averaging periods should be monthly, seasonal, or annual depending on the underlying criteria and response time of the receiving water.

4.2.4 Critical Conditions and Frequency of Excursion

Critical conditions utilized for toxic WLA models are excessively rare and transient to serve for nutrient WLAs. For example, a 1Q10 or 7Q10 streamflow reflects an averaging period (1 day or 7-day) that is much shorter than that over which nutrient impacts would occur. As discussed in section 4.2.3, quantitative nutrient goals are often set as seasonal or annual average values. It would inappropriate to set a critical condition that has a much lower duration component than the nutrient target itself. For this reason, critical conditions for nutrient WLA models would more reasonably be expressed as monthly, seasonal, or annual average conditions, depending on the timing to which the water body in question is expected to respond.

Critical conditions for nutrients would vary by water body type. In wadeable streams, the potential for algal biomass accrual is highest in lower flow, warmer periods when there is little scour of attached algae and less dilution of point source inputs (USEPA, 1999). Similarly, low flushing rates can cause low-flow years to represent the critical years in larger rivers and the riverine segments of reservoirs and estuaries.

However, for longer-retention time systems, the exact opposite pattern can be true because higher-flow years can increase nonpoint source nutrient loading and trigger algal blooms (Boesch and Brinsfield, 2000). In estuarine settings, high inputs of freshwater can also exacerbate water column stratification and hypolimnetic hypoxia (Boynton and Kemp, 2000). With these critical conditions for nutrient WLAs could be based on a typical “wet” year or “dry” year/season, depending on the system response, rather than a rare and transient 1Q10 or 7Q10 condition.

The TSD recommends a 1-in-3 year average frequency for exceedance of both acute and chronic criteria. Because criteria are developed using conservative assumptions, the TSD states that “small excursions above the criteria should not result in measureable environmental impacts”. Frequency components acknowledge that it is impracticable to guarantee that water quality criteria will never be exceeded, and that aquatic systems are resilient and can recover from toxicity events.

It can be reasonably expected that aquatic life is as resilient or more resilient to nutrient-related excursions as toxics impacts, for reasons that are rooted in the differences between nutrient and toxic impact mechanisms discussed in Section 2. For this reason, it is not recommended that nutrient WLAs be developed using frequency assumptions that are more stringent than would be used for toxics. For this reason, steady state conditions for nutrient WLA models should generally not reflect frequencies that are rarer than once in three years. As an example of a frequency component for nutrient goals, Florida’s numeric nutrient criteria are expressed as annual geometric means not to be exceeded more than once every three years (Florida DEP, 2013). This approach acknowledges that interannual climatic variability and uncontrollable meteorological events (e.g., hurricanes) affect attainability, but that extremely conservative frequency components are not appropriate for nutrient goals.

Example – Critical Streamflows

The table below provides a summary of critical low river flows calculated from historical USGS monitoring station data for 16 US rivers selected to represent a range of flow rates and climatic conditions. Comparison of calculated river statistics for alternative critical condition definitions (7Q10, 14Q5, and 30Q2) indicates significant low flow variation depending upon the statistic selected for evaluation.

On average for the data set, 14Q5 and 30Q2 flows are 27% and 96% greater than 7Q10 flows, respectively. Therefore, selection of appropriate averaging periods and recurrence intervals for critical low river flows with respect to nutrient analyses plays a key role in establishing assimilative capacity of the waterbody, because WLAs are directly proportional to the critical river flow available for mixing and assimilation of nutrients within the waterbody.

River (location)	USGS Monitoring Station	7Q10 Flow (cfs)	14Q5 Flow (cfs)	% Increase Over 7Q10	30Q2 Flow (cfs)	% Increase Over 7Q10
Chehalis River (Grand Mound, WA)	12027500	124	148	19.4%	207	66.9%
Rogue River (near Medford, OR)	14339000	871	954	9.5%	1,130	29.7%
Yakima River (near Yakima, WA)	12503000	723	798	10.4%	1,000	38.3%
Platte River (near Ashland, NE)	06801000	520	852	63.8%	1,990	282.7%
Yellowstone River (Billings, MT)	06214500	1,190	1,620	36.1%	2,260	89.9%
Colorado River (CO/UT border)	09163500	1,410	1,760	24.8%	2,550	80.9%
Klamath River (Klamath, CA)	11523000	1,250	1,460	16.8%	1,990	59.2%
Sacramento River (near Red Bluff, CA)	11378000	3,070	3,730	21.5%	5,210	69.7%
Arkansas River (near Little Rock, AR)	07263500	1,570	2,420	54.1%	5,540	252.9%
Willamette River (at Salem, OR)	14191000	3,960	4,760	20.2%	6,200	56.6%
Susguehanna River (Harrisburg, PA)	01570500	2,700	3,350	24.1%	5,540	105.2%
Snake River (Weiser, ID - WA/ID border)	13269000	6,980	8,010	14.8%	9,810	40.5%
Tennessee River (near Chattanooga, TN)	03571850	12,200	15,200	24.6%	21,100	73.0%
Missouri River (Waverly, MO)	06895500	8,340	12,800	53.5%	22,400	168.6%
Ohio River (Cincinnati, OH)	03255000	6,730	8,310	23.5%	12,900	91.7%
Columbia River (at The Dalles, OR)	14105700	75,800	84,400	11.3%	99,100	30.7%
			Average	**26.8%**	**Average**	**96.0%**

Element of Toxics-Based Permitting	1 in 3 year frequency of excursion for aquatic life protection.
Summary of USEPA Training Slide Content	Acknowledges that frequency component of nutrient criteria must be considered, but does not provide specific recommendations. Recommends that permitting agencies determine the appropriate frequency by consultation with water quality standards staff or literature.
Conclusion of Review	**Appropriate** for nutrients, if longer averaging periods are used. If shorter averaging periods (e.g., monthly) are applicable, the allowable excursion frequency may be higher.
Recommended Modifications of TSD Approach	If shorter averaging periods (e.g., monthly) are applicable, the allowable excursion frequency may be higher.

4.2.5 Mixing Zone Concepts

A mixing zone can be thought of as a limited area or volume that allows treated effluent to disperse and become diluted, while not resulting in an adverse impact to the overall receiving waterbody. USEPA defines the concept of a mixing zone in Section 4.3 of the TSD, stating that a mixing zone is an "allocated impact zone" where numeric water quality criteria can be exceeded as long as the following conditions are met:

- The mixing zone does not impair the integrity of the waterbody
- There is no lethality to aquatic life that may pass through the mixing zone
- There are no human health risks, considering likely pathways of exposure.

Other TSD recommendations for mixing zones include:

- Evaluation of the potential for bioaccumulation of toxic pollutants in fish tissue to unsafe levels
- Allowing for a zone of passage for swimming or drifting aquatic life
- Preventing impairment or blocking of critical resource areas
- Additional protections against effluents that may attract aquatic life.

As discussed in Section 2, nutrients are usually not directly managed to prevent against most of the potential impairments above. Nutrients do not bioaccumulate, and human health risks and acute toxicity are only associated with very high concentrations of particular species (e.g., nitrate). Ammonia nitrogen is usually regulated in its own right as a toxic constituent. In fact, where nutrient limits are based on protection of a downstream reservoir or estuary, near-field effects may be of little concern. However, in some settings there is the potential for discharges to cause near-field exceedances of response variables. This raises the question of whether mixing zone concepts for toxics should be applied or modified for nutrients.

USEPA describes two types of mixing zones with respect to the near-field toxicity to aquatic life and recommends sizing of these mixing zones based on the typical exposure duration. Chronic criteria must be met at the edge of (but not within) the chronic mixing zone, which is sized to protect the ecology of the waterbody as a whole. A smaller area, the acute mixing zone, is sized to prevent lethality to passing organisms. Acute criteria must be met at the edge of (but not within) the acute mixing zone. In practice, the acute and chronic mixing zones assume daily (acute) and monthly (chronic) exposure durations. Although maximum mixing zone dimensions vary by state, the chronic mixing zone generally includes a distance of several hundred feet downstream (or in a radius) from the discharge, with a percentage of that area reserved for the acute mixing zone.

In general, the authors of this review conclude that the concept of mixing is appropriate with respect to nutrients because typical waterbody impacts (i.e., excessive biological growth) resulting from the discharge of nutrients must be processed in the aquatic environment in order to have an impact. However, because nutrients do not cause direct toxicity or bioaccumulate, toxics-based limitations on mixing zone dimensions have limited value and are overly restrictive. In addition, mixing/dilution dynamics alone do not account for the total reduction of nutrient concentrations within the waterbody. The biological uptake (assimilation) of nutrients provides an additional dispersal pathway for nutrients.

4.2.5.1 Protection Against Downstream or Far-Field Impacts

Where nutrient WLAs are primarily controlled by the need to protect against downstream or far-field impacts, it will not be necessary to define mixing zones for nutrients. Rather, WLA models should assume full mix, and also take into account the non-conservative behavior of nutrients in the aquatic environment.

4.2.5.2 Protection Against Near-Field Impacts

In some permitting situations, nutrients may be regulated to prevent eutrophic effects both in the far-field and the near-field. For example, it may be desirable to prevent nuisance algal accumulations just downstream of the outfall. However, it might not always be practical to achieve concentration targets or growth-limiting concentrations in the near field, particularly in smaller streams or rivers. Just as mixing zones for toxics allow for an area of assimilation while limiting the overall impacts to the water body, permits/regulations should allow areas of assimilation/impact for nutrients. Potential equivalents to chronic and acute mixing zones could be defined as follows:

- An Assimilation Zone (AZ) where nutrient concentrations could exceed numeric criteria as long as numeric and/or narrative criteria for potential response variables are not exceeded. Definition of the AZ would potentially include defined reaches/segments of watersheds based upon site-specific evaluations or estimates of available light, presence of aquatic species, and other assimilation variables.
- A Nutrient Mixing Zone (NMZ) would be a smaller area located within the AZ where both nutrient concentrations and response variable criteria could be exceeded as long as no acute aquatic life impacts were anticipated.

Based upon the watershed level impact of nutrients and seasonal rate of assimilation, near-field toxicity-based mixing zone definitions would be overly restrictive and are not applicable to nutrients. For example, because there is no concern of lethality to passing or drifting organisms, it would not be necessary to limit AZ/NMZ dimensions based on the width or cross-section of the receiving water, and the authors conclude AZ/NMZ could be larger than acute/chronic mixing zones for toxics. In stream/river settings, it would probably be most practical to define AZ/NMZ dimensions based on the length of the stream below the outfall. Where scientifically-based numeric nutrient criteria (linked to designated uses) have been established, compliance with these criteria needs to be evaluated at the AZ and NMZ over appropriate averaging periods (e.g., seasonal or annual) to account for the inherent time scale of nutrient assimilation and observation of nutrient impacts. It is acknowledged there would be situations where protection against downstream or far-field impacts would result in more stringent limits than would be required for the AZ/NMZ alone.

These concepts merit additional consideration by regulatory agencies and stakeholders. In some cases, it might be necessary to modify state water quality standards with mixing/assimilative zones definitions that are specific to nutrients.

Element of Toxics-Based Permitting	Use of toxicity-based mixing zones.
Summary of USEPA Training Slide Content	States that “an outright prohibition on considering dilution or mixing is not likely for nutrients.”, and that “provisions allowing consideration of dilution…generally would apply to nutrients”.
Conclusion of Review:	**Inappropriate** (and unnecessary) where nutrients are managed for far-field effects. In these situations, a full mix assumption should be used. **Appropriate with modifications** if near-field effects are a concern.
Recommended Modifications of TSD Approach	Potential mixing/assimilation concepts to be explored include: • Assimilation Zone where nutrient concentrations may be exceeded as long as numeric and/or narrative criteria for potential response variables are not exceeded. • Smaller Nutrient Mixing Zone within the Assimilation Zone where both nutrient and response parameter criteria may be exceeded, as long as acute effects are anticipated and avoided.

4.2.6 Additional WLA Considerations for Nutrients

There are several additional considerations for deriving nutrient WLAs that either do not apply to toxics constituents, would be applied differently for nutrients, or are not fully addressed in the TSD or USEPA training materials. These include: (1) preferential nutrient control; (2) bioavailability; (3) seasonality; and (4) equity and cost-effectiveness.

4.2.6.1 Consideration of Preferential Nutrient Control

Another fundamental difference in WLAs for nutrients and toxics is the need to consider two parameters (nitrogen and phosphorus) jointly. Although it is recognized that some toxics can have synergistic effects, attainment of each individual toxic criterion is assumed to prevent impairments. However, responses to nutrient loading can be very different depending on the relative levels of control of nitrogen and phosphorus. For receiving waters that are co-limited by nitrogen and phosphorus, it is possible to achieve the same levels of response variables with different combinations of nitrogen and phosphorus reduction, and some combination might be much more cost-effective than others.

It has long been recognized that because some blue-green alga taxa can fix nitrogen from the atmosphere, low nitrogen-to-phosphorus ratios can favor blue-green algal dominance. For example, Smith (1983) found that blue-green algae were rare in lakes for which the N:P ratio exceeded 29:1, and Stockner and Shortreed (1988) concluded that nitrogen-fixing blue-green blooms were favored by a low N:P ratio. Over control of nitrogen (relative to phosphorus) can thus exacerbate blue-green algal blooms. It is recommended that WLA efforts that are aimed at reducing blue-green algae explicitly consider N:P ratios, and consider the benefits of preferentially controlling phosphorus.

Another potential reason to preferentially control phosphorus over nitrogen in lakes is to maintain sufficient nitrate in the water column to avoid iron-reducing or anaerobic conditions in shallow lake sediments, which tends to release phosphorus and ammonia to the water column and can exacerbate algal blooms. The best known example of this strategy is the Occoquan Reservoir in Virginia, where higher nitrate concentrations are maintained specifically to protect reservoir water quality (Cubas and others, 2014).

For receiving waters with response variables that are strongly limited by one nutrient—or where such a nutrient limitation can be imposed—the preferred strategy will be to focus controls on the limiting nutrient. Reduction of the non-limiting nutrient may require expensive controls with no tangible environmental benefits.

This scenario points out a major drawback of permitting nutrients to independently-applicable nitrogen and phosphorus concentration targets instead of response variables. Some degree of control of the non-limiting nutrient may still be warranted to protect downstream waters.

For example, Paerl (2005) demonstrated that phosphorus reductions in the freshwater (P-limited) portions of a river-estuary system actually exacerbated algal blooms in the lower (N-limited) segments, primarily because there were less algae in the upper segments to utilize nutrients, and therefore higher nutrient loads were passed to the downstream segments.

4.2.6.2 Bioavailability

USEPA has provided regulatory options (e.g., water effects ratios, chemical translators, biotic ligand model) for considering the bioavailability or site-specific effects of toxic constituents. Similarly, not all nutrients species are equally bioavailable to algae. A portion of the dissolved organic nitrogen (DON) component in many wastewater discharges is highly resistant to degradation (i.e., “recalcitrant” or “refractory”) and does not sustain algal biomass over short timeframes (Ikenberry and others, 2003). In longer retention time systems, a portion of this DON can become variable (Sedlak and Pehlivanoglu, 2006; Mulholland and others, 2007), but up to half can remain non-bioavailable (Awobamis and others, 2007).

POTWs that treat wastewater from some industrial sources have higher-than-normal refractory DON in the effluent. For these reasons, it would be recommended that guidance on nutrient permitting include options for considering the low bioavailability of select nutrient species. Options could include regulation of only the more bioavailable forms, or providing dischargers the option of demonstrating the refractory nature of a portion of their nutrient load, so that it would not count against their WLA. For example, Colorado’s Regulation 85 includes limits for total inorganic nitrogen instead of total nitrogen, and Virginia’s regulations (9VAC25-720-40) include the provision that “on a case-by-case basis, a discharger may demonstrate to the satisfaction of the board that a significant portion of the nutrients discharged by the facility is not bioavailable to aquatic life”.

4.2.6.3 Seasonality

Seasonality is not a major issue for permitting toxics constituents, which are presumed to be toxic year-round. However, in most river and stream settings, nutrients discharged during the cold weather months have little effect, and year-round limits would be unnecessary unless needed to protect downstream uses. USEPA (2007b) states that “seasonal permit limits may be acceptable if they are consistent with applicable water quality standards, and with the assumptions and requirements of the wasteload allocation of any approved TMDL.”

4.2.6.4 Consideration of Equity, Cost-Effectiveness, and Treatability Limitations

Section 4.2 of the TSD document briefly discusses methods for allocating WLAs among multiple sources of loading. Other USEPA documents provide additional guidance for distributing available assimilative capacity among sources in the context of TMDLs and WLAs. For example, the draft *Handbook for Developing Watershed TMDLs* (USEPA, 2008) cites “equitability among sources” and “feasibility of allocations” as factors that affect allocation decisions. Although these factors apply to both nutrients and toxics, they are especially important to consider for nutrients due to the large nonpoint source contributions of nutrients in many settings, and the wide variety in cost-benefits of different nutrient removal technologies.

It is recommended that WLA and LA distribution efforts for nutrients address all major sources and consider the cost-benefits of specific distribution schemes. Many agricultural nutrient reduction methods are highly cost-effective compared with point source nutrient controls (Weiland and others, 2009; Jones and others, 2010).

Although point source nutrient removal can be cost-competitive with other source reduction methods at certain treatment levels, the most stringent treatment tiers may have poor cost-benefits and ancillary detriments. For example, as part of a recent WERF study, Falk and others (2013) demonstrated that the most stringent nutrient removal tiers at POTWs achieve only marginally more nutrient removal than more moderate nutrient removal tiers, but at much higher costs.

The WERF study also concluded that the most stringent nutrient removal tiers have various ancillary detriments including higher energy use, chemical use, solids generation, and greenhouse gas emissions.

Treatability limitations at POTWs are discussed in more detail in section 3.2. In addition to concerns already discussed, extremely low nutrient WLAs can limit POTW capacity with long-term implications for economic growth. One potential outcome is an increase in unregulated septic systems, which can actually increase nitrogen loading to surface water in the long term. For these reasons, it is recommended that WLA distribution efforts for nutrients address *all* major sources and explicitly consider the cost-benefits of specific distribution schemes.

4.3 Deriving Water Quality-Based Effluent Limits

The TSD describes a limited number of statistical methods to derive water quality based effluent limits (WQBELs) from wasteload allocations (WLAs). The TSD discourages permit writers from using WLAs for toxics directly as WQBELs. The primary reason cited is that USEPA policy requires the consideration of effluent variability and sampling frequency, and how these would affect the ability to verify compliance with WLAs. The TSD's recommended method for deriving WQBELs for toxics is to identify a "target" statistical distribution that protects against excursions of the WLA. The default assumption is that effluent concentrations are lognormally distributed, although the method can be modified for data that follow other distributions. The statistical distribution is defined by its long-term average (LTA) and its coefficient of variation (CV), which represents the relative variation of a data set defined as the standard deviation divided by the mean. WQBELs for the maximum daily limit (MDL) are based on an estimate of the upper bounds (e.g., 95th or 99th percentile) of the distribution. WQBELs that require averaging (e.g., the average monthly limit or AML) are based on an upper bound of the distribution of the average values, and consider how the sampling frequency would affect the confidence in the estimate.

The TSD document discusses several variations of how this method could be applied, including:

- A two-value (acute and chronic) approach that uses a WLA derived from a steady-state model.
- A single-value approach that uses a WLA derived from a steady-state model.
- A single-value approach that uses a WLA derived from a dynamic model.

Because nutrient targets are not expressed as acute or chronic value, the two-value approach obviously does not apply to nutrients. However, USEPA and some state permitting agencies have used the single-value methods (or variations thereof) for deriving WQBELs for nutrients.

The authors of this review agree that it is appropriate to consider effluent variability when setting permit limits, both for protection of the environment (water quality-based considerations) and for avoiding unattainable limits (technology-based considerations). However, we disagree that consideration of effluent variability means that WLAs cannot be used as WQBELs for nutrients in some circumstances. In addition, there are empirical alternatives to the TSD approach that need to be considered, especially when robust datasets are available and default TSD assumptions (e.g., lognormality) break down. This subsection addresses the derivation of WQBELs in three different manners: (1) modifying the TSD's statistical approach for nutrients; (2) setting the WQBELs to the WLAs; and (3) using empirical alternatives to the TSD approach.

4.3.1 Modifying the TSD's Statistical Method for Nutrients

This subsection addresses the concept of applying the TSD's statistical approach to nutrients. Each major assumption and calculation step is discussed in subsections below.

4.3.1.1 Validity of the Lognormal Assumption

As discussed in Section 3.1, the WERF investigators (Bott and Parker, 2011) found that the nutrient species that is not well characterized by lognormal distributions are the ammonia-nitrogen data, as much of the data for daily and even monthly averages is below the minimum detection limit (MDL) and therefore is typically plotted as half the MDL, resulting in no slope for much of the distribution and a slope for the rest. This means that a lognormal distribution is not a valid approach for the analysis of ammonia data.

However, for total nutrients, there was a better fit by lognormal distributions. All the frequency distributions available from the WERF project workshop proceedings were examined for conformance to the lognormal distribution (see Table 4-1). In all but two cases, the daily data fell above the lognormal distribution line in the critical high probability range of interest. For the monthly data, the majority of cases (12) fell on or below the lognormal distribution line rather than above the line (8 cases). And on an annual basis, all the nutrient data fell on or below the lognormal distribution line.

The conclusion is that a lognormal distribution is usually appropriate for consideration in settling annual or monthly limits. The poor validity of the fit for the daily limits is perhaps not relevant, because daily limits would be inappropriate for nutrients. The caveat of the WERF project team must be reiterated here, however, in that only 36 months of data were available, and therefore the conclusions drawn here would have to be validated by examination of longer records in the future.

Table 4-1. Summary of Conformance to Log-Normal Distribution in the High Probability Range of Interest (95 to 99%); Indication is Whether Data is Above, On, or Below the Fitted Log-Normal Distribution Line

Plant	Annual[a]	Month[b]	Daily
River Oaks, FL			
TN	Below	Below	Above
Eastern WRF, FL			
TN	Below	Above	Above
Parkway, MD			
TN	Below	Above	Above
Fiesta Village, FL			
TN	Below	Below	Below
Western Branch, MD			
TN	Below	Below	Above
Scituate, MA			
TN	Below	On	Above
Truckee Meadows, NV			
TN	Below	Above	Above
Piscataway, MD			
TN	Below	Above	Below

Tahoe-Truckee, CA			
TN	Below	Above	Above
Iowa Hill, CO			
TP	Below	Above	Above
F. Wayne Hill, GA			
TP	Below	Above	Above
Cauley Creek, GA			
TP	Below	Below	Above
Clark County, NV			
TP	On	On	Above
Rock Creek, OR			
TP	na[c]	Below	Above
Blue Plains, DC			
TP	Below	Below	Below
ASA, VA			
TN	Below	Below	Above
TP	Below	Below	Above
Pinery, CO			
TP	Below	Below	Above
Kalispell, MT			
TP	On	Below	Above
Kelowna, BC			
TP	Below	Above	Above

Note: a. 12 month rolling average, b. 30 day rolling average, c. Not available because dry weather permit only

Element of Toxics-Based Permitting	Default assumption that effluent concentrations are lognormally distributed.
Summary of USEPA Training Slide Content	Describes TSD approach of assuming lognormality unless demonstrated otherwise.
Conclusion of Review:	**Appropriate** for total nutrients as a default assumption, lacking facility-specific data.
Recommended Modifications of TSD Approach	Consistent with the TSD recommendations, if facility-specific data are available, the data need to be evaluated for compliance with the lognormal assumption. The use of actual probability distributions is an alternative to the assumption of any distribution function.

4.3.1.2 Coefficients of Variation

Because CVs are an intermediate step in the fitting of the distributions, the WERF investigators did not separately report them in their report. However, CVs were subject to wide variability and resulted in values both above and below the 0.6 value used in the TSD. There were insufficient data to generalize about how the CV would change with treatment process. Therefore, use of actual distributions, whether based on the data or by fitting the data to a lognormal distribution, is the most accurate measure available at the present time.

Element of Toxics-Based Permitting	Default assumption of CV equal to 0.6.
Summary of USEPA Training Slide Content	The slides use CV in example calculations, but do not explicitly address the default assumption that CV = 0.6.
Conclusion of Review:	**Appropriate** for total nutrients as a default assumption, lacking facility-specific data. Actual CVs are highly variable between facilities, and there were insufficient data to generalize about how the CV would change with treatment process or averaging period.
Recommended Modifications of TSD Approach	Consistent with the TSD recommendations, if facility-specific data are available, they should be used to calculate a facility-specific CV. The use of actual probability distributions is an alternative to the need to calculate a CV.

4.3.1.3 Calculation of the Long-Term Average

The TSD presents equations for calculating the LTA of the "target" distribution from the WLA (see Step 2, Box 5-2 of the TSD). These include an equation for calculation of separate LTAs to protect against excursions of the acute criteria and chronic criteria. Because nutrients are not generally controlled for acute impacts, neither the equation for the acute LTA nor the associated multiplication factors (see Table 5-1 of the TSD) are appropriate for nutrients. Similarly, the given equation for calculating the chronic LTA assumes that the underlying criterion and WLA has a 4-day averaging period. As discussed in section 4.2.3, significantly longer averaging periods are usually appropriate for nutrients.

Therefore, as with the acute LTA equation, neither chronic LTA equation nor the associated multiplication factors (see Table 5-1 of the TSD) are valid for nutrients. For longer averaging periods, the LTA will be much closer to the WLA than is generally warranted for toxics.

If using the TSD method, the permit writer must modify the equation for calculating LTA based on the appropriate averaging period of the WLA (growing season, annual, etc.). Specifically, the standard deviation associated with the 4-day averages in the chronic LTA equation must be replaced with the standard deviation of the appropriate averaging period. The modified equation is as follows:

$$LTA = WLA \cdot e^{[0.5\sigma_n{}^2 - z\sigma_n]}$$

Where σ_n^2 is the variance of the effluent concentration at the appropriate n-day averaging period, and may be calculated as:

$$\sigma_n{}^2 = \ln(\frac{CV^2}{n} + 1)$$

Probability Basis of the LTA Calculation: The z-value in the LTA calculation equation is set based on the desired probability basis. The probability basis presented in the TSD ranges from the 95th percentile (z= 1.645) to the 99th percentile (z=2.326). The higher the probability basis, the lower the LTA that will result from a given WLA. For toxics, USEPA recommends a default 99th percentile probability basis of the 99th percentile for calculation of the LTA. This is a highly conservative value and is based on the concept that the WLA for toxics will almost never be exceeded. However, there are differences between toxics and nutrient impacts that will lead to the use of a less conservative probability basis for nutrients in many circumstances. As discussed in section 2, the impacts associated with nutrients can be more indirect, gradational, or aesthetics-based than toxics impacts.

The selection of a 95th percentile probability basis is still a relatively conservative approach, and would still result in an LTA that is less than the WLA, but would acknowledge (where appropriate) that the rare exceedance of the nutrient WLA is not expected to cause lethality to aquatic life or harm to human health. This judgment would depend upon the nature of the receiving water and nutrient impacts that the WLA was derived to prevent.

Element of Toxics-Based Permitting	Method for calculating long-term average.
Summary of USEPA Training Slide Content	States that longer averaging periods are sometimes appropriate for nutrients, and provides permitting examples.
Conclusion of Review:	**Appropriate for LTAc only with modification** for nutrients.
Recommended Modifications of TSD Approach	The correct averaging periods must be used to calculate the LTA from the WLA. Probability bases should be less conservative than those used for toxics.

4.3.1.4 Calculation of AML/MDL

Federal regulations [40 CFR 122.45(d)] require that, unless impracticable, NPDES permit limits be expressed as both average monthly and maximum daily values for dischargers other than POTWs, and as average weekly and average monthly limits for POTWs. However, the expression of WQBELs for nutrients as daily limits is unnecessary and inappropriate due to the lack of short-term/acute effects. For the same reasons, the expression of WQBELs for nutrients as weekly limits will also be inappropriate for systems with longer retention times, or those that otherwise do not respond to nutrient loadings on a weekly time scale. Section 4.2.3 contains an extensive discussion of why longer averaging periods are appropriate for nutrient WLAs rather than the averaging period associated with WLAs for toxics. These same considerations should be use to (1) rule out shorter averaging periods for permit limits as impracticable; and (2) select appropriate averaging periods for limits that are tied to the duration components of the criteria or the time frame to which the system is expected to respond to nutrient inputs.

There is ample precedent for the use of longer averaging periods for nutrient WQBELs. For the protection of the Chesapeake Bay, the USEPA (2004) made a formal determination that it was “impracticable” to express effluent limitations as daily maximum, weekly average, or monthly average”, and that annual permit limits were appropriate. Similarly, the Wisconsin DNR (2012) prepared a formal justification for the use of monthly, growing season (6-month), or annual averaging periods for NPDES limits, depending in part on the residence time of the system.

Similarly, in Massachusetts, NPDES permits for POTWs discharging to inland waters contain seasonal limits for P, expressed as a 60 day rolling average during the April thru October season.

With the need for longer averaging periods in mind, the TSD method for calculating and expressing permits limits must be modified. The equations for calculating MDLs are not relevant because MDLs are not appropriate for nutrients. The equation for calculating the AML would be appropriate if the other assumptions of the TSD method (e.g., lognormal distribution) were correct and a monthly permit limit was warranted. However, for longer averaging periods, it would be necessary to modify the interpretation of the equation. Specifically, a limit for an n-day average would be calculated using the following equation:

$$n - day\ average\ limit = LTA \cdot e^{[z\sigma_n - 0.5{\sigma_n}^2]}$$

Where n is the number of samples collected over the entire n-day averaging period, not the number of samples collected per month.

Probability Basis for Calculating Time-Averaging Limits: As with calculation of the LTA, the calculation of permit limits requires the selection of z-values that is tied to the desired probability basis (typically the 95th - 99th percentile). However, unlike with the LTA equation, a higher probability basis results in a less conservative result. In other words, a permit limit calculated using a 99th percentile probability will be higher (less stringent) than a permit limit calculated using a 95th percentile probability, all other factors being equal. As stated in the TSD, USEPA default recommendation is to use a 99th percentile probability basis with the MDL, and a 95th probability basis with the AML. In general, the authors' opinion is that the 97th-99th percentile is the appropriate range of probability basis to use with time-averaged limits for nutrients. This range acknowledges that permit limits are intended to represent the upper limit of acceptable plant performance and that, as compared with a toxics limit, results in a somewhat less conservative assumption regarding the frequency of exceedance that is appropriate for nutrients.

Element of Toxics-Based Permitting	Calculation of MDLs and AMLs
Summary of USEPA Training Slide Content	States that longer averaging periods are sometimes appropriate for nutrients, and provides permitting examples.
Conclusion of Review	MDLs are **inappropriate** for nutrients. AMLs may also be inappropriate, depending on the applicable averaging period of the WLA.
Recommended Modifications of TSD Approach	Limits must be calculated and express the appropriate averaging periods, which may be monthly, seasonal, or annual.

Example – Effect of Averaging Period

The Rocky River WWTP has been assigned a WLA with a concentration basis of 1 mg/L, properly interpreted as a 30-day average concentration based on the underlying criterion and WLA model. The coefficient of variation (CV) is 0.6 and the sampling frequency is 4 times per month. The LTA is calculated using a 95% probability basis, and the MDL is calculated using a 99th percentile basis.

If the WLA were misinterpreted as a 4-day average (e.g., based on chronic toxicity),using the LTA equation in section 4.3.1.3 that accounts for the 4-day averaging period, the resulting LTA would be 0.644 mg/L, and the AML woul d be 1.22 mg/L.

If the WLA were properly interpreted as a 30-day average, the resulting LTA would be 0.841 mg/L, and the AML would be 1.59 mg/L —a thirty percent difference.

LTA at 95% probability basis; AML at 99% probability basis			
Statistic	Units	Value Using 4-day avg.	Value Using 30-day avg.
Z95		1.645	1.645
Z99		2.326	2.326
WLA_c	mg/L	1.0	1.0
CV		0.6	0.6
σ^2		0.3075	0.3075
σ		0.5545	0.5545
n-day average for WLA		4	30
$\sigma^2_{n\text{-day avg}}$		0.08618	0.01193
$\sigma_{n\text{-day avg}}$		0.29356	0.10922
LTA_c	mg/L	0.644	0.841
n	samples/mo.	4	4
$\sigma^2_{n\text{-sample}}$		0.08618	0.08618
$\sigma_{n\text{-sample}}$		0.29356	0.29356
AML	mg/L	1.22	1.59

4.3.1.5 Mass-Based vs. Concentration Based Limits

Federal regulations [40 CFR 122.45(f)] require that NPDES limits be expressed in terms of mass, with a few exceptions for parameters such as pH. In the TSD document, the USEPA also recommends that permit limits be expressed in units of concentration for receiving waters with less than 100-fold dilution. The cited purpose is to prevent elevated receiving water concentrations of toxic substances in circumstances where the effluent concentration is not greatly reduced by mixing/dilution. An extreme case of this circumstance would be an effluent-dependent water.

The same reasoning that is cited above for toxics would also apply to nutrients if WLAs were directed based on valid, near-field concentration targets. However, mass limits for nutrients would usually be sufficient where WLAs were developed on the basis of loading to a downstream lake, reservoir, estuary, or large river segment.

Section 4.2.1 of this report discusses different situations where WLAs should be based on concentration targets vs. load-response linkages. Much of this same information is relevant to whether concentration-based limits are necessary in any given permitting situation.

In the *NPDES Permit Writer's Manual* (USEPA, 2010c), USEPA states that concentration-based limits could be considered to discourage a reduction in treatment efficiency during low effluent flow periods. However, this is primarily a technology-based consideration rather than a water quality-based consideration, and so is not directly relevant to the derivation of WQBELs. Concentration-based limits could complicate to nutrient trading programs, which typically operate on a mass basis. Nutrient trading has the potential to greatly reduce the overall costs of achieving nutrient reduction goals (Jones and others, 2010) and has been successfully performed in multiple states. Concentration-based limits can also provide a disincentive to water conservation programs. For this reason, the authors of this review recommend that WQBELs for nutrients be expressed in terms of mass only unless concentration-based limits have been demonstrated to be needed to protect designated uses.

4.3.2 Implicit Consideration of Effluent Variability: Using WLAs as WQBELs

Compared with the TSD's statistical approach for calculating WQBELs, a simpler approach is to set WQBELs to the WLAs at the appropriate averaging period. The TSD discourages this approach for toxics out of concern that it is insufficiently conservative; i.e., that compliance with a monthly average (30-day) limit could still be associated with exceedance of the chronic (4-day) WLA. This is primarily a concern when the averaging period of the limit is much longer than the averaging period of the underlying water quality criterion and WLA. However, this is rarely the case for nutrients, and in fact the opposite is more likely to be true. Nutrient criteria and WLA are more likely to have longer averaging periods, similar in magnitude to the averaging periods of the limits themselves. For longer-averaged WLAs, the use of WLAs as WQBELs is equally *or more* conservative than the TSD approach.

This is demonstrated by the inset example, in which a WWTP's annual average limit is shown to be *higher* than the WLA, if the TSD approach were used with the appropriate averaging periods. The LTA will always be lower than the WLA when using the TSD equations. However, as illustrated on Figure 5-8 of the TSD, the limits for any averaging period can be higher or lower than the WLA, depending on the CV, sampling frequency, and probability bases. As the averaging period of the WLA increases to 30-days and longer, the AML will usually be higher than the WLA. Hence, setting monthly, seasonal, or annual WQBELs to the WLA tends to be a conservative approach. In addition, as the averaging period of the WLA and sample number increase, the LTA becomes closer to the WLA, such that there is little difference between the TSD approach and simply setting the WQBEL to the WLA.

Other Sources of Conservatism that Justify Implicit Consideration of Effluent Variability: WLA models often have various levels of conservativeness already built in. Water quality criteria themselves tend to be conservative, and federal regulations require that TMDLs include a margin of safety. As discussed in section 4.2.2.1, steady-state WLA models typically assume the coincidence of critical receiving water conditions with maximum point source discharges. Even most dynamic WLA models place all point sources at maximum discharge and loading levels simultaneously, which was cited as a major contributor to the margin of safety of the Chesapeake Bay nutrient TMDL (USEPA, 2010a). This is highly unlikely to ever occur in reality.

Just as all POTWs within a watershed are unlikely to be simultaneously discharging at maximum design flows, it is highly unlikely that they would simultaneously experience the upper bounds of their "random" effluent variability.

Rather, the random variabilities at different facilities would tend to cancel each other out. In this manner, the variability in the total nutrient loading would be lower than the variability in loads from individual facilities.

State Precedents: The use of WLAs as WQBELs has well-established precedents for successful management of nutrients in high-profile watersheds, including rivers, reservoirs, and estuaries. Examples include:

- Implementation of the Chesapeake Bay nutrient TMDL in **Pennsylvania** and **Virginia**. For example, Virginia DEQ's permitting guidance states that "The annual loading limit for total nitrogen and total phosphorus for these facilities is what is referred to…as waste load allocations." (Virginia DEQ, 2007)
- Implementation of point source nutrient controls in the Long Island Sound, **Connecticut**. Connecticut's General Permit for Nitrogen Discharges states that "A permittee shall be in compliance with it annual discharge limits of this general permit if…the POTW's annual mass loading of total nitrogen is less than or equal to [the WLA]."
- **North Carolina's** approach for implementing point nutrient controls in multiple basins, include both river-estuary systems (Tar-Pamlico River, Neuse River) and reservoirs (Jordan Lake, Falls Lake).
- **Minnesota's** general watershed permit for phosphorus in the Minnesota River Basin, which sets limits as 5-month mass limits derived from a TMDL.
- **Nevada's** approach for permitting phosphorus loads to the Truckee River. For example, the monthly average phosphorus mass limit for the Truckee River Water Reclamation Facility is expressed as the facility's WLA from a TMDL.
- **Colorado** has implemented a TMDL for nitrate/total inorganic nitrogen by including the monthly mass loading allocations from the TMDL directly as the permit limitations.

In summary, the concerns expressed in the TSD regarding the direct use of WLAs as WQBELs generally do not apply to nutrients. If anything, the use of WLAs as WQBELs may be too conservative in some situations. However, when using longer averaging periods and relatively high sampling frequencies, the two approaches will provide similar limits, and the direct use of WLAs and WQBELs should be considered a viable alternative.

Element of Toxics-Based Permitting	Discouragement from using WLAs as WQBELs
Summary of USEPA Training Slide Content	Topic not explicitly discussed, and most examples sue TSD approach for considering effluent variability. However, one example shows the use of a WLA as an annual average WQBEL.
Conclusion of Review	**Inappropriate** for nutrients. Due to longer averaging periods and higher sample numbers, setting WQBELs to WLAs is a conservative approach that implicitly considers effluent variability.
Recommended Modifications of TSD Approach	The setting of WQBELs to WLAs can simplify permitting and trading calculations while providing defensible permit limits.

Example – Comparison of TSD Approach with Setting WQBEL to the WLA

The Sandy Shoals WWTP has been assigned a WLA based on 8.0 mg/L total nitrogen as an annual average. The CV is 0.3 and the sampling frequency is weekly. The LTA is calculated using a 95% probability basis, and the average annual limit is calculated using a 99th percentile basis. Properly interpreting the WLA as an annual average, the resulting LTA is 7.8 mg/L, and the annual average limit is 8.6 mg/L. Hence, at this long averaging period, the TSD approach gives a similar but slightly less conservative average annual limit than simply setting the average annual limit to 8.0 mg/L.

LTA at 95% probability basis; averaged limit at 99% probability basis		
Statistic	**Units**	**Value**
Z95		1.645
Z99		2.326
WLA	mg/L	8.0
CV		0.3
σ^2		0.0862
σ		0.2936
n-day average for WLA	days	365
$\sigma^2_{\text{n-day avg}}$		0.00025
$\sigma_{\text{n-day avg}}$		0.01570
LTA_c	mg/L	7.80
n	samples/year	52
$\sigma^2_{\text{n-sample}}$		0.00173
$\sigma_{\text{n-sample}}$		0.04158
Average annual limit	mg/L	8.58

4.3.3 Alternative to TSD Method: Empirical Distribution Functions

Although there is a wide variety of potential methods for explicitly considering effluent variability, the two basic categories most relevant to deriving WQBELS are: (1) use of theoretical probability distribution functions; and (2) use of actual probability distributions. The TSD approach represents the former approach. The validity of this method rests largely in the validity of the assumptions regarding the data distribution and its parameters (e.g., the CV), and the application of the appropriate averaging periods.

As discussed in section 4.3.1.1, the validity of the assumption of lognormality varies greatly by facility, and departures from lognormality sometimes occur in the region most applicable to permitting (i.e., the upper tail of the distribution), depending on averaging period.

In contrast to the use of probability distribution functions, actual probability distributions require no assumptions regarding the distribution of the effluent data. Rather, the empirical distribution function is based on the actual distribution of the data as determined from historical monitoring data. The function can be displayed as a concentration-probability plot (e.g., Figure 3-1 in Section 3) displays the percentage of values that were less than or equal to a range of observed concentrations. Such plots can be developed for any averaging period of interest, such as daily values, monthly averages, etc.

Using the TSD method, the "target" distribution for developing WQBELs does not usually represent the observed distribution; rather it represents a distribution that would achieve the WLA. If process changes are required to meet the WLA, the target distribution may be lower in overall magnitude than the observed distribution, and could potentially have a different variability as well. This requires an estimate of the effluent distribution after the plant complies with the limit. Unless means are available to predict the future CV, practitioners of the TSD method typically assume that the CV (or in some cases, the ratio of the upper bound percentile to the expected value) remains constant.

Similar assumptions are required to derive WQBELs from an empirical distribution function. Unless WQBELs are to be based on the existing condition, the observed empirical distribution function would have to be shifted upward or (more commonly) downward on the concentration –probability plot to achieve the WLA. A simple assumption for making this shift is that the ratios of the percentiles (e.g., the ratio of the 95th percentile to the 50th percentile) of the distribution would remain the same. Under this assumption, the shift would be accomplished by changing the observed values by the same ratio. Specific steps for deriving a WQBEL from the concentration-probability plot would be as follows:

a. Express the WLA as a target concentration at the appropriate design flow, with an averaging period appropriate to the receiving water body and manner in which the WLA was derived.

b. Develop the concentration-probability plot based on observed data averaged using the appropriate averaging period. The Weibull probability associated with each concentration can be calculated as:

$$P = \left(\frac{rank}{n+1}\right)$$

Where *P* is the probability, *n* is the number of data points in the set, and *rank* is the rank of the concentration in the data set.

c. Choose a probability basis that corresponds to a low rate of exceedance of the WLA. Generally, this should be in the 90th-99th percentile range, and can be lower for longer averaging periods.

d. Reduce or increase the unaveraged (daily) observed concentration values by a constant percentage until the concentration-probability plot (of averaged values) is shifted such that the target WLA corresponds to the selected probability basis.

e. Ensure that shifted values do not decrease below a technologically-achievable level. This can be accomplished be setting a technologically-based "floor" (see for example Section 3.2 or Bott and Parker, 2011) for a lower limit set at the 95 percentile monthly value to the amount that the observed values can be shifted downward.

f. The shifted daily values represent the "target" distribution. They can be used to calculate concentration-probability plots for any appropriate averaging period, and WQBELs can be set based on the upper bounds (e.g., 95th-99th percentile) of the probability plots.

The calculations described above can easily be performed in a spreadsheet. As with the TSD approach, it would be necessary to evaluate the resulting limits for technological feasibility, and ensure that that the WLA did not result in unachievable WQBELs. If the WQBELs are unachievable, this should cause permitting agencies to reevaluate the WLA and probability bases, or potentially take an alternative (non-WQBEL) permitting approach.

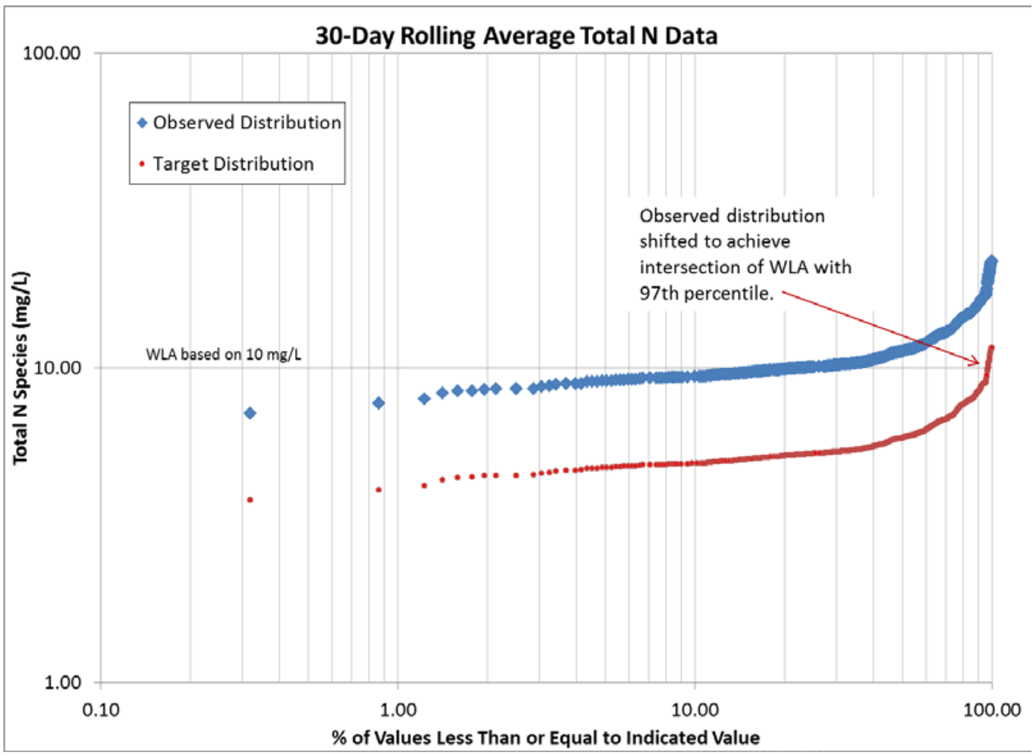
Advantages: The primary advantage of the empirical distribution method is that it requires no default assumption of a probability distribution, and can be more accurate for situations where the target distribution can be reasonably expected to be similar to the observed distribution except in magnitude. The advantage is greater for data distributions that do not conform well to the lognormal distribution or other common distributions. Unlike the TSD approach, the empirical approach can directly consider a lower limit to treatability (see step #5 above) in the derivation of the target distribution. Some users may also find the expression of the WQBELs as percentiles of the target distribution as more intuitive than limits derived using the TSD approach.

Disadvantages: The primary disadvantage of the empirical distribution method is that it requires a large historical dataset. For use in a permitting situation, it would generally be recommended to have at least 36 months of frequent monitoring to characterize the distribution. As with the TSD approach, this approach requires assumptions regarding the variability of the future (target) distribution, based on the observed distribution. If the treatment process will change significantly, there would be less confidence that the future distribution will be similar in shape to the observed distribution, and the empirical method thus might not be more accurate the TSD method.

Note that data from the WERF study of effluent variability (Bott and Parker, 2011) summarized in Section 4.3.1.1 showed that in 8 out of 20 cases examined, the actual data fell above the fitted lognormal distribution for plants that had very low effluent limits. Where this is the case, WQBELs derived from an actual probability distribution function would actually be more stringent than derived from the TSD default method. This is because, compared to the default lognormal assumption, it would be necessary to shift the observed probability plot down more to have the target WLA coincide with the probability basis of the limit. Therefore, the empirical approach does not necessarily provide less stringent limits than the TSD approach, and the primary advantage is in more accurate limits for distributions that depart from lognormal assumptions.

Example – Calculating Limits from an Empirical Data Distribution

The Rocky River WWTP receives a WLA for total nitrogen based on concentration of 10 mg/L, properly interpreted as a 30-day rolling average. The WWTP has a long-term monitoring dataset that shows common exceedance of this limitation. A 47% reduction in the observed (daily) values would result in 30-day rolling averages that meet the WLA 97% of the time (see graph below). The upper (blue) data set represents the empirical distribution of observed data over an appropriate averaging period for nutrients. The lower (red) data set was shifted to meet a WLA of 10 mg/L as a 30-day rolling average with 97 percent probability, which would correspond to only about once exceedance every three years. The shifted data set was generated by multiplying the individual daily data by a factor of 0.53.



Section 5

Summary and Recommendations

The results of this review demonstrate that the procedures for developing WLAs and WQBELs for nutrients should differ from toxics permitting in multiple and profound ways. These differences are partially rooted in fundamental differences in the mechanisms of action for nutrient and toxic-related impairments, and the high degree of variability in how water bodies respond to nutrients. Unlike the dose-response effects expected from toxics, nutrient effects are often better characterized as indirect, gradational, and water body-specific. Differences between nutrients and toxics point toward factors such as longer averaging periods, different critical conditions, and a preference for load-response predictions over the use of default concentration targets. Unlike most toxic parameters, nutrients are contributed by almost every point and nonpoint source on the landscape, requiring a close consideration of broad watershed based control strategies that incorporate technological feasibility, equity, cost-effectiveness, and overall environmental effects.

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5.1 Summary of TSD Review

The core of this review was an evaluation of the appropriateness of elements of the TSD approach for developing nutrient WLAs and WQBELs. Results of the review are summarized in Table 5-1, which categorizes TSD elements as:

- Rarely appropriate for nutrients.
- Appropriate for nutrient permitting in certain circumstances, or with recommended modifications to the TSD method.
- Usually appropriate for nutrient permitting.

At the broadest level, the approach for developing WQBELs for nutrients and toxics would be similar: develop WLAs using the best available science, and develop WQBELs from the WLAs in a manner that addresses effluent variability. However, as demonstrated in Table 5-1, there are relatively few aspects of the TSD approach and assumptions that merit direct application to nutrients and many TSD elements that require reexamination and modification for application to nutrients. The TSD's basic approach for conducting a quantitative reasonable potential analysis for toxics is not applicable to nutrients. Nutrient WLA derivation methods should vary significantly from those used for toxics, with a preference for load-response linkages where in-stream concentration targets are not available or reliable for predicting response variables. Critical conditions, frequency of excursion, averaging periods, and application of mixing/dilution concepts could also differ substantially between toxics and nutrients.

It is the authors' view that, after a science-based nutrient WLA is derived, the TSD's basic statistical framework for calculating nutrient WQBELs from WLAs could be successfully modified for nutrients in many circumstances. Key modifications should include verification of log normality, application of longer averaging periods, and selection of probability bases that reflect a level of conservativeness that is appropriate to nutrients. The direct use of WLAs as WQBELs and empirical distribution functions represent viable alternatives to the TSD method for considering effluent variability.

Table 5-1. Summary of TSD Component Review Results		
Toxics-Based Permitting Component	**Result of Evaluation**	**Explanation**
Assumption of dose-response; acute and chronic concepts	○	Nutrient effects are indirect, gradational, and often water body-specific.
Human health concepts	○	Nutrients do not bioaccumulate, are not carcinogenic, and are not directly toxic except for certain species at relatively high levels.
Focus on concentration targets	○ / ⊙	Valid concentration targets may be available in some settings; for others, load is more meaningful.
Excursion frequency for protection of aquatic life	⊙	1-in-3 year frequency may be appropriate if longer averaging periods are used.
Critical conditions	⊙	Should be more common and less transient that 1Q10/7Q10 conditions used for toxics.
Reasonable potential analysis based on 95^{th} to 99^{th} percentile	○	Nutrient impacts not controlled by short-term spikes.
Deriving WLAs from steady state models	⊙	Appropriate in limited circumstances. Steady state condition must be modified to reflect appropriate critical condition.
Deriving WLAs from dynamic models that link designated uses to nutrient concentrations	●	Preferred for complex water bodies.
Consider downstream effects	●	Nutrient WQBELs must protect downstream uses.
Consider variability in effluent characteristics when setting WQBELs	⊙	Effluent variability should be considered implicitly or explicitly.
Assumption of log normality	⊙	Verify with facility-specific data.
Method for calculation of the long-term average	⊙	Modify to reflect longer averaging periods of WLA. Probability basis can be lowerthan with toxics.
Expressing limits as maximum daily limits	○	Daily time frames not meaningful for nutrients.
Expressing limits as longer averaging periods	⊙	Monthly, seasonal, or annual limits are appropriate, depending on the receiving water body dynamics.
Relevance of # samples per averaging period	⊙	Using longer averaging periods, the sensitivity of limits to sample number is less. Can be considered implicitly.
Mixing zone concepts	⊙	Use a full mix when protecting against far field effects. Near field effects could merit alternative mixing/assimilation zone concepts.

Key: ○ Rarely appropriate for nutrient permitting.

⊙ Appropriate for nutrient permitting in certain circumstances, with recommended modifications to the TSD method.

● Usually appropriate for nutrient permitting.

In addition to considering the appropriateness of TSD components for nutrients, the reviewers also identified factors that should be considered for nutrient permitting but are not fully addressed in the TSD, or otherwise commonly considered for toxics permitting. These include alternative methods for deriving WLAs, including load-response linkages and bioconfirmation-based methods.

An example of the latter approach would be deriving WLAs from existing conditions where biological conditions are favorable. Because nonpoint sources are a major source of nutrients in most watersheds, it is even more important than with toxics to ensure that all major sources are addressed in an equitable fashion. The high costs of nutrient removal should prompt consideration of cost-benefit, treatability limitations, ancillary detriments, and options for preferentially controlling one nutrient (phosphorus or nitrogen) to attain water quality goals in a more cost-effective manner.

5.2 Recommendations on a Nutrient Permitting Framework

The TSD review was largely a science-based technical review process of evaluating individual components of permitting methods. Having completed this detailed review, it is helpful to consider how the results would constructively inform elements of viable nutrient permitting frameworks. The authors recognize that permitting is the responsibility of both USEPA and states, and nutrient permitting approaches would vary based on hydrologic settings and state-specific regulations. However, it is possible to identify broad concepts and approaches that would aid the development of sound WLAs and WQBELs in most settings. Many of these concepts require departures from the toxics paradigm to one degree or the other. The recommendations can be placed into three basic categories: (1) methods for deriving WLAs; (2) methods for calculating WQBELs; and (3) treatability considerations.

5.2.1 Recommendations on Deriving Wasteload Allocations

High-level recommendations for deriving nutrient WLAs include the following:

> *Recommendation #1: Recognize load-response linkages as an alternative to the use of concentration targets for deriving WLAs*.
>
> The toxics permitting paradigm is focused on attaining in-stream concentration targets. For reasons discussed in this review, the nutrient concentration targets may not be useful or appropriate for deriving WLAs in many settings. The use of default concentration targets such as USEPA's 2000 ecoregional criteria or Gold Book values do not allow consideration of how specific water bodies would respond to nutrient inputs, and could be either over- or under-protective depending on setting. Load-response linkages will often represent a superior approach for deriving WLAs, and specifically for considering water body-specific responses. If load-response linkages are available, NNC should not be considered necessary for permitting. A special case of where load-response linkages should take precedence over concentration targets is where a technically-valid TMDL or other restoration plan has already established load-response linkages and WLAs.
>
> *Recommendation #2: Provide options for use of bioconfirmation to inform WLAs*.
>
> Several states (ME, OH, FL) have made progress on bioconfirmation approaches for implementation of nutrient standards, whereby response variables take precedence over nutrient concentrations for determination of impairment. The presence of favorable response variables should prevent the lowering of WLAs from existing conditions. Depending on other factors, the finding of elevated nutrient concentrations but favorable response variables should prompt (1) a determination of no reasonable potential of impairment from existing discharges; or (2) WLAs based on existing conditions or antidegradation considerations; or (3) derivation of site-specific nutrient criteria.
>
> USEPA (2013b) has endorsed aspects of the bioconfirmation approach, but USEPA's "guiding principles" for bioconfirmation still recommend the use of default nutrient concentration targets for permitting. This would negate much of the benefit of bioconfirmation by assuming that a single set of nutrient concentration targets is appropriate for protecting designated uses. Under USEPA's approach, bioconfirmation would benefit assessment but not permitting.

Recommendation #3: Consider preferential nutrient controls to attain desired responses:

Preferential nutrient controls have no direct parallel with toxics. For reasons discussed in this report, the WLA derivation process should consider whether preferential controls of either N or P would attain response variables more cost-effectively, or be superior for blue-green algal control. In contrast, independent focus on both N and P concentration targets could lead to costly nutrient removal efforts with little additional environmental benefit.

Recommendation #4: Use critical conditions and frequency components that are tailored to nutrient responses.

Due to the temporal, spatial, and mechanistic aspects of how water bodies respond to nutrient inputs, default critical conditions and averaging periods for toxics will generally not be appropriate for nutrients. Basing nutrient WLAs on very rare hydrologic conditions (e.g., 1Q10 or 7Q10 flows) will usually result in unnecessarily low WLAs. Rather, critical conditions should represent more common (if conservative) seasonal conditions.

Recommendation #5: Nutrients should usually be permitted assuming full mix and accounting for non-conservative behavior.

The size of mixing zones for toxics is limited by factors such as the need to prevent lethality to passing organisms. These considerations do not apply to nutrients, which are regulated to manage impacts over broad spatial scales and frequently are based on control of downstream impacts. For this reason, nutrients should generally be granted a full mix assumption if managed on the basis of far-field effects. Near field effects should prompt exploration of alternative mixing/assimilation concepts.

Recommendation #6: Use watershed-based permitting approaches that consider equity, cost-effectiveness, and ancillary effects directly in the WLA derivation process.

Unlike toxics, nutrients are derived from almost every part of the landscape. Although toxics controls can sometimes focus on point sources, this is not generally the case for nutrients. WLA derivation methods should ensure that all major sources (point and nonpoint) are addressed in an equitable manner. Cost-benefit analysis should be used to inform how nutrient reduction can be achieved in the most economical manner, and avoid wasteful and inefficient public investments. This should include consideration of the high incremental costs, low incremental benefits, and ancillary detriments (chemical use, energy use, solids generation, GHG emissions, etc.) of the most stringent point source nutrient removal tiers.

5.2.2 Recommendations on Calculating WQBELs

High-level recommendations for calculating nutrient WQBELs include the following:

Recommendation #7: Use the appropriate averaging periods for deriving and expressing limits.

The appropriate averaging period is one the most fundamental differences between nutrients and toxics that can affect WQBELs and attainability. Under the TSD method, the averaging period is for calculation of LTAs and WQBELs, and also in considering how to express the limit. Averaging periods for nutrients should be tailored to the receiving water body response, but will always be higher than the 1-hour and 4-day averaging periods used for acute and chronic toxics criteria. In some cases, growing season or annual averaging periods are appropriate, particularly for receiving water bodies with longer retention times. Permit limits should be expressed using averaging periods that are no shorter than the timeframe that is meaningful for nutrient water quality response.

Recommendation #8: Verify or modify assumptions regarding lognormality and coefficient of variation using facility-specific data.

Research referenced in the review has revealed that effluent nutrient data often depart from the TSD's default assumption of lognormality in the region of most relevance to permitting (i.e., the upper percentiles). Coefficients of variation for nutrients are highly variable, depending on technology employed and contributing sources but can be lower or higher than the default assumption of 0.6 at many facilities. Where available, facility-specific data should be used.

Recommendation #9: Choose probability bases that reflect a reasonable level of conservativeness for nutrients.

Several steps of the WQBEL calculation procedures require the selection of percentiles that represent the upper bound of the target distribution. These values can have an important control on the stringency of the resulting limits. Because nutrients do not generally cause acute or human health impacts, and many aquatic systems can adapt to changes in nutrient supply, the probability basis for nutrient limits should be lower than with toxics.

Recommendation #10: Consider using WLAs as WQBELs.

The TSD's discouragement from using WLAs as WQBELs is primarily rooted in a concern that short-term criteria could be exceeded. When using the longer averaging period and higher sample numbers available with nutrients, this concern no longer applies. Use of WLAs as WQBELs can simplify watershed trading mechanisms, and so should be a preferred permitting option.

Recommendation #11: Consider the empirical alternative to the TSD approach for calculating WQBELs.

The review presents the use of the actual probability distribution as an alternative to the TSD's basic approach for calculating WQBELs. This method requires a robust historical monitoring data set, but can be more accurate for data distributions that do not conform well to the lognormal distribution or other common distributions. The empirical approach is especially applicable if WQBELs are to be based on existing conditions, or if the target distribution can be reasonably assumed to be similar to the observed distribution except in magnitude.

5.2.3 Recommendations on Considering Treatability Limitations

Recommendation #12: WQBELs should not be set to levels that cannot be reliably attained.

The focus of the TSD and this review is the derivation of water-quality based effluent limits, not technology-based effluent limits. However, research referenced in this report demonstrates that even very well-run, state-of-the-art facilities experience treatability limitations and variability in effluent quality. Permit writers should evaluate WQBELs in light of treatability limitations. If WQBELs would not be attainable, or would result in unavoidable, frequent exceedances, permit writers should consider options such as (a) increasing the WLA; (b) selecting different probability bases; (c) increasing the averaging period; (d) regulating only bioavailable nutrient forms; or (e) using non-WQBEL permitting approaches.

Figure 5-1 presents an example of a permitting framework for nutrients that incorporates many of the recommendations above. In this permitting framework, agencies would have several different approaches for developing WLAs, depending on the circumstance. This tiered approach is similar to Florida DEP's (2013) proposed approach for implementing nutrient standards, in that alternative methods would take precedence over the use of default concentration targets for permitting. These alternatives include the use of existing TMDLs, basing limits on existing conditions for biologically healthy water bodies, and application of load-response models.

Nutrient-specific permitting considerations such as equity and treatability would be considered jointly with WLA development. If none of these options were viable—or if they would result in unattainable WLAs with uncertain benefits—agencies might take a non-WQBEL permitting approach that combines attainable limits with adaptive management.

The permitting framework also includes three different options for calculating WQBELs from WLAs. Options discussed in this report include use a modified TSD approach, setting WQBELS to WLAs, or using actual probability distributions. Properly applied/modified for nutrients, all three of these methods are capable of giving similar WQBELs for the same WLA. Dissimilarities between nutrients and toxics should be considered at every permitting step, from reasonable potential analysis to WLA to WQBEL calculation. It is hoped that this review and recommendations will lead to a useful discussion of scientifically and technically appropriate approaches for nutrients, and aid in the development of appropriate nutrient permitting frameworks at the federal and state level.

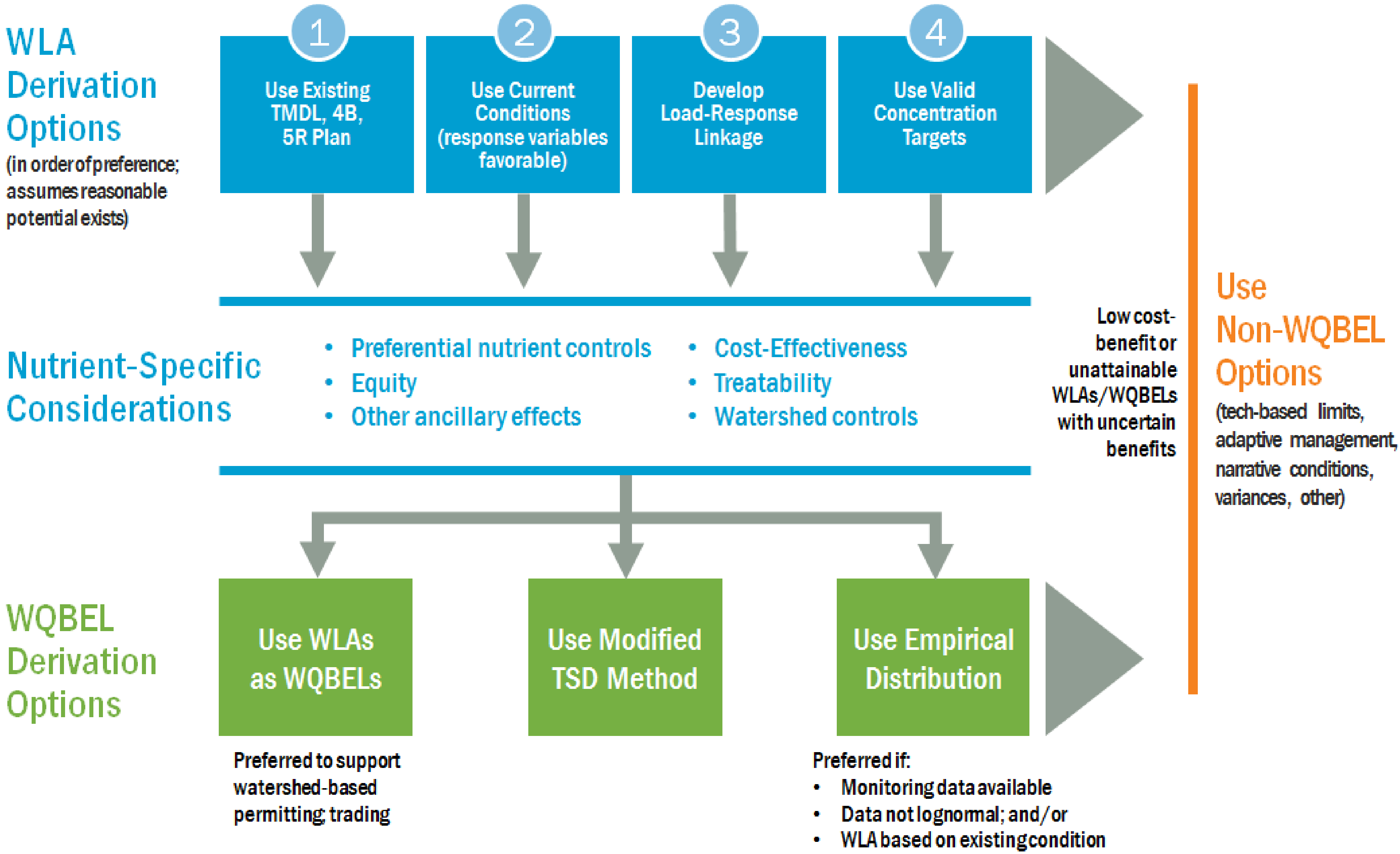

Figure 5-1. Schematic of a potential framework for deriving nutrient limits.

Section 6

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Appendix A: Summary of Results of Review of TSD Permitting Components

Summary of Results of Review of TSD Permitting Components

TSD Topic	Report Section	Summary of USEPA Training Slide Content	Conclusion of Review
Qualitative or semi-quantitative methods for determining whether a NPDES limit is needed.	4.1.1	Discusses options for performing qualitative RPA, many of which would be better characterized as semi-quantitative because they use effluent and water quality data.	**Appropriate with modification** for nutrients. Quantitative RPA is preferred, and purely qualitative RPA methods are discouraged for nutrients. However, semi-quantitative RPA can represent a reasonable approach where calibrated models are not available. Semi-quantitative RPA approaches for nutrients need to be objective, reproducible, consider assimilative capacity, take into account all factors, and be subject to public review/comment. Various characteristics of the receiving water should be considered to evaluate assimilative capacity, including current impairment status. Screening-level modeling approaches have potential to provide valuable information, lacking calibrated models. A purely qualitative approach is not recommended.
Quantitative method for determining whether a NPDES limit is needed; specifically, the comparison of upper-bound estimates to water quality criteria.	4.1.2	Describes a method very similar to the approach recommended for toxics, using nutrient concentration targets. Discusses options for selecting default concentration targets where state NNC are lacking, such as the 1986 Gold Book or USEPA ecoregional criteria. Mentions alternative to use an “indicator parameter”. Discusses modeling options in broad terms.	**Inappropriate** for nutrients in situations where valid concentration targets (linked to designed uses) are lacking. **Appropriate with modification** for nutrients if valid concentration targets (linked to designated uses) are available. If valid concentration targets are available, TSD methods need to be modified to estimate the upper bound of the effluent concentration using the appropriate averaging period. Time-averaged concentrations will usually be significantly lower than the upper bounds estimated from the unmodified TSD approach. If valid concentrations are not available, it is recommended to use calibrated load-response models or semi-quantitative screening approaches for RPA.
Assumption of dose-response and availability of valid concentration targets (linked	4.2.1	Assumes that nutrient concentration targets are available or can be selected by permit writer. Discusses options for selecting default concentration targets where state NNC are lacking, such as the	**Appropriate with modification** for some systems, and **inappropriate** for others. One of the high-level decisions in any permitting application is the determination of whether valid concentration targets are available, or whether the system is best managed using load-response linkages. Load-based goals will generally be more useful for permitting in large, complex systems such as estuaries, coastal areas, and linked river-reservoir systems, and more conducive to support water

to designated uses).		1986 Gold Book or USEPA ecoregional criteria. Mentions alternatives to use an "indicator parameter" or site-specific criteria. Also mentions the option of basing limits on "stressor-response relationships relating downstream response variables or narrative criteria to upstream nitrogen and phosphorus concentrations or loadings."	quality trading programs. If valid concentration targets are available, frequency/duration components need to be selected appropriately. If system responds primarily to loads, WLAs need to be based on load-response linkage instead of concentration targets.
Derivation of wasteload allocations from steady-state or dynamic models.	4.2.2	Discusses modeling options in broad terms. Examples use steady-state models. No explicit discussion of when a steady-state or dynamic model should be selected	Model-based WLA methods are **appropriate with modification** for nutrients but must be tailored to the hydrologic system and eutrophic responses of interest. Allocating WLAs for nutrients also requires additional considerations such as equity, technological treatment limitations, and cost-effectiveness (see section 4.2.6). Recommended modifications: • Model to response variables unless valid numeric nutrient concentrations targets are available. • Select models based on hydrologic setting, response variables of interest, data availability, etc. Use models capable of predicting the non-conservative behavior of nutrients. • Utilize longer averaging periods (e.g., seasonal, annual) for WLAs • Use dynamic WLA models instead of steady-state WLA models for moderately-complex to complex eutrophic responses (e.g., complex river, reservoir, or estuarine settings). • Consider equity, technological treatment limitations, and cost-effectiveness in deriving WLAs (see section 4.2.6).
1-hour to 4-day averaging periods of criteria/WLA.	4.2.3	Acknowledges that longer averaging periods are often appropriate for nutrients. Discusses criteria for determining when seasonal or annual averaging periods are applicable.	**Inappropriate** for nutrients, which affect water bodies over longer time scales. Nutrient criteria/WLA averaging periods should be monthly, seasonal, or annual depending on the underlying criteria and response time of the receiving water.
1 in 3 year frequency of excursion for	4.2.4	Acknowledges that frequency component of nutrient criteria must be considered. Recommends that	**Appropriate** for nutrients, if longer averaging periods are used. If shorter averaging periods (e.g., monthly) are applicable, the allowable excursion frequency may be higher. If shorter averaging periods (e.g., monthly) are

aquatic life protection		permitting agencies determine the appropriate frequency by consultation with water quality standards staff or literature.	applicable, the allowable excursion frequency may be higher.
Use of toxicity-based mixing zones	4.2.5	States that "an outright prohibition on considering dilution or mixing is not likely for nutrients."; and that "provisions allowing consideration of dilution…generally would apply to nutrients".	**Inappropriate** (and unnecessary) where nutrients are managed for far-field effects. In these situations, a full mix assumption must be used. **Appropriate with modification** if near-field effects are a concern. Potential mixing/assimilation concepts to be explored include: • Assimilation Zone where nutrient concentrations may be exceeded as long as numeric and/or narrative criteria for potential response variables are not exceeded. • Smaller Nutrient Mixing Zone within the Assimilation Zone where both nutrient and response parameter criteria may be exceeded, as long as acute effects are anticipated and avoided.
Default assumption that effluent concentrations are lognormally distributed.	4.3.1.1	Describes TSD approach of assuming lognormality unless demonstrated otherwise.	**Appropriate** for total nutrients as a default assumption, lacking facility-specific data. Consistent with the TSD recommendations, if facility-specific data are available, the data need to be evaluated for compliance with the lognormal assumption. The use of actual probability distributions is an alternative to the assumption of any distribution function.
Default assumption of CV equal to 0.6.	4.3.1.2	The slides use CV in example calculations, but do not explicitly address the default assumption that CV = 0.6.	**Appropriate** for total nutrients as a default assumption, lacking facility-specific data. Actual CVs are highly variable between facilities, and there were insufficient data to generalize about how the CV would change with treatment process or averaging period. Consistent with the TSD recommendations, if facility-specific data are available, they should be used to calculate a facility-specific CV. The use of actual probability distributions is an alternative to the need to calculate a CV.
Method for calculating long-term average.	4.3.1.3	States that longer averaging periods are sometimes appropriate for nutrients, and provides permitting examples.	**Appropriate with modification** for nutrients. The correct averaging periods must be used to calculate the LTA from the WLA. Probability bases should be less conservative than those used for toxics.
Calculation of MDLs and AMLs	4.3.1.4	States that longer averaging periods are sometimes appropriate for nutrients, and provides	MDLs are **inappropriate** for nutrients. AMLs may also be **inappropriate**, depending on the applicable averaging period of the WLA. Limits must be calculated and express the appropriate averaging periods, which may be

		permitting examples.	monthly, seasonal, or annual.
Discouragement from using WLAs as WQBELs	4.3.2	Topic not explicitly discussed, and most examples sue TSD approach for considering effluent variability. However, one example shows the use of a WLA as an annual average WQBEL.	**Inappropriate for nutrients**. Due to longer averaging periods and higher sample numbers, setting WQBELs to WLAs is a conservative approach that implicitly considers effluent variability. The setting of WQBELs to WLAs can simplify permitting and trading calculations while providing defensible permit limits.

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