

**Report of the  
Academic Advisory Committee  
To  
Virginia Department of Environmental Quality –  
Freshwater Nutrient Criteria**

**Submitted to:**

Division of Water Quality Programs  
Virginia Department of Environmental Quality

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## Table of Contents

Acknowledgements	p. iii
Acronyms and Abbreviations	iv
Executive Summary	1
<i><u>Part I: Introduction and Policy Context</u></i>	
Introduction	3
Conceptual Basis	7
Near-term Recommended Approach	12
Longer-term Recommended Approach	17
<i><u>Part II: Responses to DEQ Questions</u></i>	
Approach	19
Form	27
Regionalization	40
Classification	41
Prioritization & Coverage	47
Inventory of Existing Data	48
Planned Data Collection	49
Data Needs	49
<i><u>Appendices</u></i>	
A: Lakes and Reservoirs Data Analysis Plan	51
B: Potential Threshold Values for Rivers and Streams from Various Sources	61
C: Influence of Nutrient Levels on Fish Populations in Lakes and Constructed Impoundments	65
References	69
Members of the 2003-2004 Academic Advisory Committee	75

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## Acronyms and Abbreviations

**AAC:** Academic Advisory Committee  
**AFDM:** ash free dry mass  
**CFR:** Code of Federal Regulations  
**CWA:** Clean Water Act  
**DCR:** Virginia Department of Conservation and Recreation  
**DEQ:** Virginia Department of Environmental Quality  
**DO:** dissolved oxygen  
**ECM:** extracellular metabolites  
**E:H:** epilimnetic volume to hypolimnetic volume  
**EPA:** U.S. Environmental Protection Agency  
**Fe:** iron  
**ft:** feet  
**ha:** hectare  
**HCO<sub>3</sub>:** hydrogen carbonate  
**kg:** kilogram  
**L:** liter  
**m:** meter  
**µg/L:** micrograms per liter (1 µg/L = 0.001 mg/L)  
**mg/L:** milligrams per liter (1 mg/L = 1,000 µg/L)  
**mg/m<sup>2</sup>:** milligrams per square meter  
**mg/m<sup>3</sup>:** milligrams per cubic meter  
**n:** number of observations in a sample (sample size)  
**N:** nitrogen  
**NASQAN:** USGS National Stream Quality Accounting Network  
**NAWQA:** USGS National Water-Quality Assessment  
**NES:** National Eutrophic Survey  
**NH<sub>4</sub><sup>+</sup>:** ammonium  
**NO<sub>3</sub><sup>-</sup>:** nitrate  
**P:** phosphorus  
**PO<sub>4</sub><sup>3-</sup>:** phosphate  
**r:** coefficient of correlation for a sample  
**r<sup>2</sup>:** coefficient of determination for a sample  
**S:** sulfur  
**SD:** Secchi depth  
**SWCB:** State Water Control Board  
**TAC:** Technical Advisory Committee  
**TN:** total nitrogen  
**TP:** total phosphorous  
**TSI:** trophic state indices  
**TSS:** total suspended solids  
**UAA:** use attainability analysis  
**USGS:** United States Geological Survey  
**VDGIF:** Virginia Department of Game and Inland Fisheries

## Executive Summary

This report is provided to the Virginia Department of Environmental Quality (DEQ) by the Academic Advisory Committee (AAC) for the purpose of suggesting approaches to the development of nutrient criteria for the state's freshwaters. EPA has required that all states develop nutrient criteria, and Virginia DEQ is in the process of responding to that directive. This report focuses on criteria development for lakes and reservoirs, and for freshwater streams and rivers. The report provides general guidance as well as responses to specific questions posed by DEQ to the AAC. Methods used to prepare the report include a review of scientific literature, the application of professional judgment by the committee members, and preliminary analyses of data sources described in Appendix A. This document is a summary of the AAC's efforts to date and is presented as an interim report in anticipation of continuing activity.

Given the definition of "criteria" in the Clean Water Act and its implementing regulations, the committee recommends that DEQ base its criteria development process upon the concept of suitability for designated uses. In Virginia, all waters are designated for the following uses: recreational uses, (e.g., swimming and boating); the propagation and growth of a balanced, indigenous population of aquatic life; wildlife; and the production of edible and marketable natural resources (e.g., fish and shellfish). Some waters have additional uses, such as for public drinking water. The committee can foresee potential conflicts in setting nutrient criteria that are designed to meet all of the designated uses, because the classification of over enrichment depends on the water-quality requirements of the designated use being protected.

The committee recommends that the natural lakes and constructed impoundments be treated separately in the criteria development process. The committee recommends that the two natural lakes, Mountain Lake and Lake Drummond, have individual nutrient criteria developed because they are different from the constructed impoundments and from each other. It may be possible to utilize ecoregion and water-body type specific criteria developed by neighboring states for these two lakes if the neighboring states contain natural lakes with similar characteristics and have conducted an appropriate criteria development process.

In this report, the AAC primarily focused its attention on the 100+ constructed impoundments in the state. The AAC believes the constructed impoundments should be classified based on the types of fisheries that they support, and possibly (pending results of data analysis) based on morphometric features that influence nutrient-algal relationships and/or fish population response. In the absence of fish population data that represent a number of impoundments and are comparable across impoundments, the committee proposes using the recreational fishery status, as rated by the Virginia Department of Game and Inland Fisheries' biologists, as an indicator of the impoundments' suitability for aquatic life. Given the limited number of high-nutrient impoundments with multiple water-quality observations, available data may not be adequate to yield defensible criteria estimates with high levels of statistical confidence. However, even if the data are not statistically defensible, the data from waters with successful fisheries could still be useful as reference values. User perception surveys,

if designed, administered, and analyzed in a scientifically valid manner, would be an appropriate mechanism for assessing suitability for recreational uses.

For streams and rivers, the committee recommends that criteria be defined to represent levels of algal biomass that impair the designated uses. Criteria development should focus on periphytic algal biomass (as represented by an indicator, such as associated chlorophyll *a*) in the majority of the state's streams, while planktonic algal biomass would be an appropriate focus in larger (i.e., deeper than wadeable), slow-moving streams. The committee has not yet reached a consensus regarding a process that could be applied to define criteria levels for these indicators but recommends against application of a "relatively undisturbed reference" or a percentile-distribution reference approach. Because most of Virginia's surface waters drain into nutrient sensitive estuaries, downstream loading effects should be considered in developing nutrient criteria for rivers and streams.

The committee also recommends that investigation of nutrient-algal biomass relationships be considered as an integral component of the criteria development process because nutrient inflows both control algal biomass and are subject to direct management controls. If such relationships can be defined with acceptable levels of precision and statistical confidence, they could be applied by DEQ in defining numeric criteria as nutrient concentrations. There are two primary advantages of such expressions, compared to expressing criteria solely as chlorophyll *a* levels: (1) It would include a capability to monitor a greater portion of the state's waters for criteria compliance with DEQ's limited monitoring resources, and (2) It would provide a capability to prescribe management actions, as responses to violations, with a higher level of precision and with greater confidence that prescribed actions would be sufficient to relieve criteria violations.

The committee recommends that DEQ take actions necessary to establish nutrient criteria in a fashion that meets scheduling requirements defined by EPA, because failure to do so could allow EPA implementation of EPA guidance criteria, which in the committee's view are inappropriate for Virginia. Considering the wide variety of physiographic, landscape development, and water-body conditions that occur within Virginia and the complexity of ecological processes governing nutrient effects, the committee is concerned that the nutrient criteria resulting from the current process, even if appropriate for conditions that occur commonly throughout the state, may prove to be inappropriate when applied in water-body conditions that are unique, unusual, or uncommon. Therefore, the committee also encourages DEQ to establish a systematic process for continued evaluation and refinement of the criteria after implementation. When nutrient criteria violations occur, the resultant TMDL process should include site-specific evaluations to determine whether or not the designated use is defined appropriately and whether or not an impairment of suitability for the designated use does, in fact, exist. As a corresponding component of this evaluation and refinement process, DEQ should also establish a procedure that would systematically seek to determine whether designated use impairments might be occurring in situations that are not associated with violations of numeric criteria.

# Part I. Introduction and Policy Context

## Introduction

The U.S. Environmental Protection Agency (EPA) requires all states to develop criteria to protect waters from impairment by nutrient enrichment. In this report, the AAC addresses general nutrient criteria development issues that face the Virginia Department of Environmental Quality (DEQ), responds to specific questions posed by DEQ, and provides additional background information that we hope will be of value to DEQ in the nutrient criteria development process.

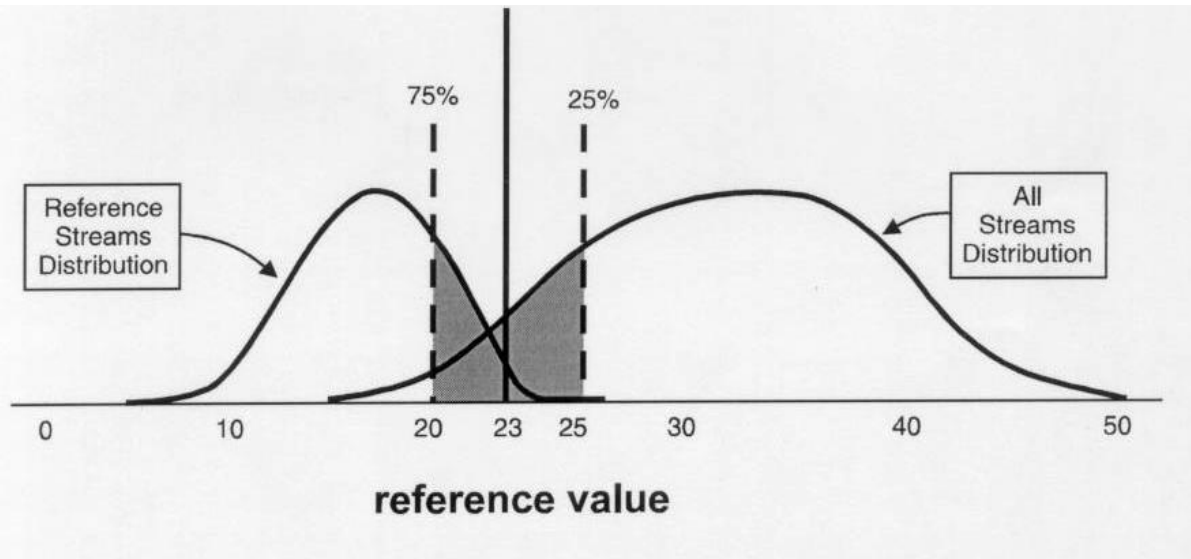
### EPA Nutrient Criteria Guidance

In 2000, the EPA issued to states guidance nutrient criteria for U.S. waters, a description of the process it used to develop those guidance criteria, and a description of processes that may be used by states for developing nutrient criteria (U.S. EPA 2000a-g). The EPA's guidance requires that the criteria be numeric and based on methods described by EPA or other scientifically defensible methods (U.S. EPA 1998). Also, the EPA made reference to such criteria being surrogates for designated uses.

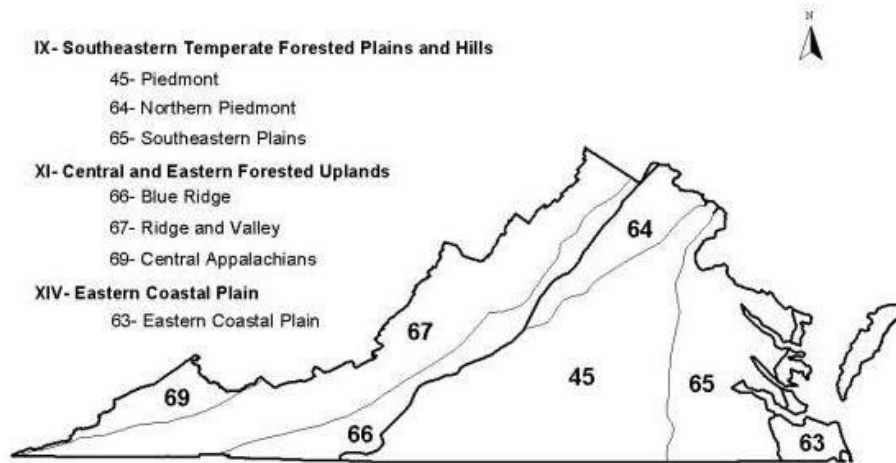
In association with its guidance to the states, EPA developed guidance nutrient criteria for streams and rivers, and for lakes and reservoirs. For states that fail to develop their own criteria using methods that are satisfactory to the EPA, the EPA has stated the intent to implement its guidance nutrient criteria.

EPA derived its guidance criteria by analyzing national water quality data for the 1990-98 period from Legacy Storet, EPA regions, U.S. Geological Survey (USGS) (NASQAN and NAWQA), and other sources. To develop these guidance criteria, the EPA aggregated the data being analyzed by water body, calculated a median for each specific water body, and then derived a median of these values for each ecoregion and water-body type (rivers and streams, lakes and reservoirs). The 25<sup>th</sup> percentiles of the water-body medians for each water-body type within each ecoregion were designated as guidance criteria.

The logic for selecting the 25<sup>th</sup> percentile is based on the identification of "relatively undisturbed" and "least impacted" reference sites. EPA recommends such reference sites as an appropriate basis for setting criteria. In ecoregion-based studies, EPA compared the distribution of nutrient levels from all sites within a water-body type to the reference sites for that water-body type and found that the 25<sup>th</sup> percentile of all the medians within the water-body type generally corresponded with the 75<sup>th</sup> percentile of the medians from the reference sites of that water-body type (Figure 1). Based on these studies, EPA defined guidance nutrient criteria as the 25<sup>th</sup> percentiles of the medians from all sites for each water-body type within each U.S. ecoregion (Figure 2, Table 1). EPA also analyzed the data to define the 25<sup>th</sup>-percentile reference values by subregion (Table 2).



**Figure 1.** Illustration of EPA concept regarding relationship of water quality values in reference conditions to values in all water bodies of the same type (U.S. EPA, 2000g; Figure 8). The caption reads: “Selecting reference values for total phosphorous concentration ( $\mu\text{g/L}$ ) using percentiles from reference streams and total stream populations.” The horizontal axis is total phosphorous ( $\mu\text{g/L}$ ). In this example, 20  $\mu\text{g/L}$  represents the 75<sup>th</sup> percentile of all reference streams while 25  $\mu\text{g/L}$  represents the 25<sup>th</sup> percentile of the “all streams” distribution. Therefore, 23  $\mu\text{g/L}$  is selected as the reference value and, by implication, as the TP criterion.



**Figure 2.** EPA ecoregions occurring within Virginia

**Table 1.** EPA nutrient criteria guidance for freshwaters in Level III ecoregions that contain regions of Virginia.

Ecoregion <sup>a</sup>	TP ( $\mu\text{g/L}$ )	TN ( $\text{mg/L}$ )	Chlorophyll <i>a</i> ( $\mu\text{g/L}$ )	Turbidity NTU	Secchi (m)
<i>Rivers and Streams</i>					
IX	36.56	0.692	0.930	7.02	n/a
XI	10.00	0.305	1.613	2.30	n/a
XIV	31.25	0.710	3.750	1.94	n/a
<i>Lakes and Reservoirs:</i>					
IX	20.00	0.358	5.18	n/a	1.53
XI	8.00	0.458	2.79	n/a	2.86
XIV	17.50	1.270	3.35	n/a	0.79

<sup>a</sup> see Figure 1.

**Table 2.** Reference conditions<sup>a</sup> for Level III and Level IV ecoregions containing Virginia components

EPA Ecoregions and Subregions <sup>b</sup>	EPA Region's Extent	TP ( $\mu\text{g/L}$ )		TN ( $\text{mg/L}$ )		Chlorophyll <i>a</i> ( $\mu\text{g/L}$ )		Turbidity (NTU)	Secchi Depth (m)
		R&S	L&R	R&S	L&R	R&S	L&R	R&S	L&R
IX. Southeastern temperate forested plains and hills	PA - TX	36.6	20	0.69	0.36	0.93	4.93	7.02	1.53
45: Piedmont	VA - AL	30	22.5	0.62	0.30	3.49	4.51	5.71	1.66
64: Northern Piedmont	NJ - VA	40	45	2.23	0.82	1.21	n/a	2.83	1.54
65: South-eastern Plains	MD - MS	22.5	10	0.62	0.35	0.05	5.13	6.20	2.04
XI. Central and eastern forested uplands	PA - AL; MO - OK	10	8	0.31	0.46	1.61	2.79	2.30	2.86
66: Blue Ridge	PA - GA	7.1	5	0.28	0.12	1	1.35	2.00	4.37
67: Ridge and Valley	PA - AL	10	17.5	0.21	0.38	1.06	3.28	2.40	2.10
69: Central Appalachians	PA - AL	7.6	5	0.50	0.59	n/a	1.75	2.18	3.36
XIV. Eastern Coastal Plain	ME - GA	31.3	17.5	0.71	1.27	3.75	3.35	1.94	0.79
63: Eastern Coastal Plain	DE - GA	52.5	n/a	0.87	n/a	3.75	n/a	3.89	n/a

<sup>a</sup> 25<sup>th</sup> percentile of medians for all sites within a water-body type, with data evaluated by EPA, 1990 – 1998. R&S = rivers and streams; L&R = lakes and reservoirs.

<sup>b</sup> See Figure 1.

Source: US EPA, 2000a-e.

## Nutrient Management in Virginia

In September of 2003, DEQ requested that the Academic Advisory Committee (AAC) provide advice and assistance in its effort to comply with EPA's requirements for developing nutrient criteria. This document is a summary of the AAC's efforts to date, and is presented as an interim report in anticipation of continuing activity. This document provides general guidance, as well as responses to specific questions posed to the AAC by DEQ.

As context, in 1986, the Virginia State Water Control Board (SWCB) appointed a Technical Advisory Committee (TAC) to assist in the development of nutrient criteria for the state's waters. Although the TAC did recommend threshold levels for defining nutrient impairment, the SWCB did not adopt those recommendations as enforceable criteria. DEQ currently uses the TAC recommendations (Table 3) as screening values for identifying "nutrient enriched waters," but it does not define impairments solely on the basis of these screening values.

**Table 3.** Virginia DEQ nutrient screening values for freshwaters (current).

Water Body Type	TP (µg/L)	TN (mg/L)	Chlorophyll <i>a</i> (µg/L)	Dissolved Oxygen
Flowing waters	100 - 200	No standard	Narrative standard <sup>a</sup>	24 hour fluctuation > <sup>1</sup> / <sub>3</sub> DO saturation
Freshwater lakes	50	No standard	25 monthly average; 50 maximum	Narrative standard <sup>a</sup>
Tidal freshwater	No standard, monitor only	No standard	120% of background	Standard related to background chlorophyll <i>a</i>
Estuarine	No standard, monitor only	No standard	120% of background	Standard related to background chlorophyll <i>a</i>

<sup>a</sup> For further details, see Virginia DEQ 2004, p. 41-43.

## **Conceptual Basis**

The requirement to develop nutrient criteria guidance highlights an important policy issue: How should the presence or absence of "clean water" be determined? On what basis should impairment be judged? The Clean Water Act (CWA) and its implementing regulations include language that can be interpreted to answer those questions. It is the committee's view that the statute and its regulations give rise to conflicting interpretations for nutrients, in part because the ecological role of nutrients differs in a fundamental manner from those water pollutants that can be expected to have toxic effects when present at concentrations that are elevated over natural background.

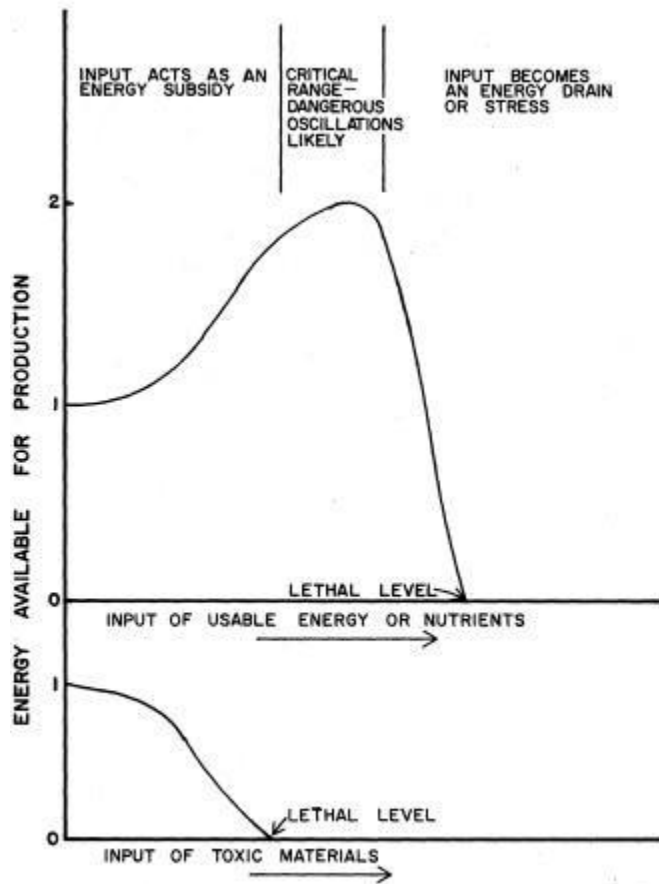
## Nutrients' Ecological Roles

The availability of nitrogen and/or phosphorous limits photosynthesis in most terrestrial and aquatic ecosystems, which means that nitrogen and/or phosphorous

inputs to such system can be expected to increase the rate of primary production by photosynthetic organisms (Smith et al. 1999). Nutrients are transmitted up food chains by consumer organisms that eliminate excess nutrients as components of waste products. While urban centers can and generally do collect and treat human wastes, similar systems are not in place to collect excess nutrients associated with the extensive managed landscapes that are commonly associated with human activities, including agricultural production systems. A great variety of human systems tend to concentrate and/or disperse nutrients on the landscape, including fossil energy combustion.

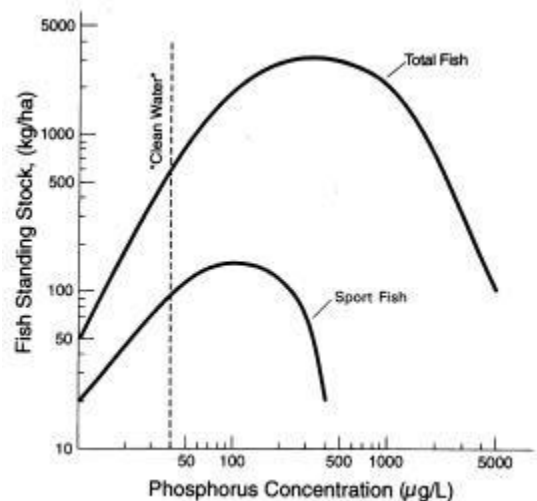
In moist, temperate regions such as Virginia, forests – mature ecosystems whose efficiency in cycling of the nutrients is well known – occupy the undisturbed landscape. As landscapes are developed for human support and habitation, forests are commonly replaced by herbaceous vegetation systems (such as cropland, pastureland, lawns, and landscaping) that require management practices that often include nutrient inputs. In an ecological sense, herbaceous systems can be described as “young” or “immature” ecosystems because if they are not managed for persistence in humid climates such as Virginia, they will revert to woody vegetation and eventually to forests. The efficiency with which forests are able to utilize and cycle nutrients contributes to these systems’ competitive advantage over unmanaged landscapes, and thus to the succession process. Using one set of terms, nutrient cycles in young ecosystems tend to be “loose,” and these cycles “tighten” as the systems mature (Odum 1969). As a result of these processes, streams draining forested systems generally carry nutrients in low concentrations, relative to streams draining landscapes dominated by other types of vegetative cover.

Although EPA’s nutrient-criteria guidance is based on an assumption that essential nutrients at concentrations above levels characteristic of a “relatively undisturbed reference” act as “pollutants,” nutrients differ fundamentally from other types of water contaminants. In fact, applying the term “pollutant” to low-level, but higher-than-reference, concentrations of N and P would be in contrast to the ecological literature. For example, ecologist Eugene Odum contrasts two types of “pollution:” degradable organics (including nutrients) and non-degradable toxic inputs (Figure 3). Although the CWA itself does not make a distinction between the two types of pollution, a reading of the Act finds the term “pollutant” used consistently in contexts that indicate such substance to be harmful or toxic. Increasing inputs of a toxic pollutant leads directly to degradation of system function, but low-level inputs of degradable organic inputs act “as an energy subsidy” (quote from Odum 1971). Only at higher levels (over enrichment) do the degradable organics become “an energy drain or stress” (again quoting from Odum 1971). Nutrient inputs can be expected to act in a manner comparable to the energy subsidy described by Odum (1971). Data gathered by Ney (1996) demonstrates this effect for southeastern reservoirs (Figure 4). Nutrients create problems through over enrichment but not by acting as direct toxic inputs. They cannot be presumed to have a toxic effect simply by virtue of being elevated above concentrations that may be observed at a “relatively undisturbed” reference. This property of nutrients is in contrast with many other water pollutants, which are capable of exerting low-level ecotoxic effects, even when elevated slightly above natural background.



**Figure 3.** Schematic model of the effects of two types of pollution – degradable organic (top) and non-degradable toxic (below). [From Odum 1971, p. 435].

**Figure 4.** Generalized relation of total fish and sport fish standing stock with total phosphorous concentration in temperate latitude reservoirs. Standing stock values are representative of southeastern U.S. reservoirs to 100  $\mu\text{g/L}$  total P, while standing stocks at higher P concentrations are hypothetical. The vertical line labeled as “clean water” represents a TP concentration associated with water clarity that could be considered as minimally acceptable for contact recreational use and is an approximate value. The “clean water” representation is conceptual and is not reproduced here for the purpose of suggesting a specific TP criterion value [from Ney 1996].



## Policy Issues

The Code of Federal Regulations (CFR) defines the term *criteria* as “elements of State water quality standards, expressed as constituent concentrations, levels, or narrative statements, representing a quality of water that supports a particular use. When criteria are met, water quality will generally protect the designated use” [40 CFR 131.3(b)]. It is clear from this language that water quality standard setting for any water body type (river, stream, lake, and reservoir) requires a definition of the uses to be achieved and the establishment of criteria that are credible surrogates for the water bodies’ suitability for those uses.

This language implies that determinations of water-body impairment status should occur after assigning the designated uses. These uses are described in various ways in the Clean Water Act. Section 102(a) states that “it is the national goal that, wherever attainable, an interim goal of water quality which provides for the protection and propagation of fish, shellfish, and wildlife and provides for recreation in and on the water ...” shall be achieved. This phrase is the basis for the common description of the CWA as supporting “fishable and swimmable” waters. The term “designated use” is used at other points in the Act, including Section 303: “...Such revised or new water quality standard shall consist of the designated uses of the navigable waters involved and the water quality criteria for such waters based upon such uses ... [shall] protect the public health or welfare, ... shall be established taking into consideration their use and value for public water supplies, propagation of fish and wildlife, recreational purposes, and agricultural, industrial, and other purposes, ... their use and value for navigation.”

In contrast to the above, EPA’s nutrient-criteria guidance documentation seeks to judge nutrient impairment through a comparison of each water body to “reference” conditions. The reference condition is found in comparable water bodies that are “relatively undisturbed” or the “least impacted” of all available water bodies of that type.

The reference technique is commonly used by biologists for evaluating the species distributions of biotic communities at study sites. In such studies, community composition is measured at a relatively undisturbed reference site to define a baseline condition, a community structure that can be considered to be “attainable” within a natural system that is similar in most ways to the study site and against which the study-site community can be compared (Hughes et al. 1986, Reynoldson et al. 1997). One reason why this technique is widely used is because biologists do not have methods available to predict the type of community that would or should occur under a given type of “natural” conditions, in the absence of human influence. In such studies, community composition is commonly expressed using a variety of metrics, such as total number of taxa, number of taxa within functional groups, proportion of total individuals occurring within taxonomic or functional groups, and so forth. The scientist conducting such studies is able to use his or her professional expertise to consider a variety of metrics and associated factors, including comparison of study site metrics to those at the reference, in evaluating the community. Reference sites are also used to determine background levels of substances that occur naturally in the environment at low concentrations (such as metals) for the purpose of comparison to measured levels at comparable sites that have been affected by human influence.

In applying the reference concept to criteria development for nutrients, EPA is advancing an approach that it developed and implemented during the 1990s when it encouraged the states to develop biological monitoring programs for water resources. With this approach, biological monitoring is used as a supplement to physical/chemical monitoring, and as a means for defining “biological integrity.” Biological monitoring at relatively undisturbed (or least impacted) reference sites, for the purpose of comparison to study sites, is a component of EPA’s recommended biological monitoring approach. Additionally, the biological monitoring data are incorporated into both the standard setting and assessment process for defining impairments (U.S. EPA 1991).

The legislative history of the Clean Water Act does indicate that congressional intent was to define “physical, chemical, and biological integrity” as a condition that is similar to the ecosystem’s “natural” (language used by the House committee report) or “pristine” (Senate report language) ecosystem state (Adler 2003). U.S. EPA (1991) defines “biological integrity” as “the condition of the aquatic community inhabiting unimpaired water bodies of a specific habitat as measured by community structure and function.” This definition was developed by an EPA-led process that involved the scientific community in the early 1980s (Davis 1995). As noted specifically by U.S. EPA (1991), the CWA’s Section 102(a) describes the “fishable / swimmable” [sic] goal as “interim,” suggesting that the ultimate use to be achieved is aquatic life support. U.S. EPA (1991), consistent with the legislative history, states, “Biological criteria can be quantitatively developed by identifying unimpaired or least-impacted reference waters that operationally define the best attainable conditions.” EPA further recommends that the states use the ecoregion concept to define reference waters.

In applying the “reference condition” concept to nutrient criteria development, EPA is extending an approach currently applied in biological assessments to chemical parameters. Inherent in this application is the assumption that any detectable difference of nutrient concentration from the reference is undesirable.

Both U.S. EPA technical documentation (2000f, 2000g) and academic literature (e.g., Dodds and Welch 2000) emphasize designated use impairments as the underlying basis for the need to establish nutrient criteria. For example, EPA states that “Nutrient enrichment frequently ranks as one of the top sources of water resource impairment. Systems are impaired when water quality fails to meet designated use criteria,” and the subsequent text describes a series of mechanisms by which water resources are impaired by over enrichment (U.S. EPA 2000g, Chapter 1). Yet, EPA has defined a process for developing nutrient criteria that emphasizes detection of differences compared to relatively undisturbed reference conditions that does not include a requirement for evidence of designated-use impairment.

### Designated Uses

Capabilities to serve designated uses are not measured directly but instead are assessed using surrogates called criteria. Some uses are readily understandable, such as drinking water safety, and criteria to represent a water body’s suitability for that use can be defined. Some uses are more open to interpretation such as “fishable,” and associated criteria may be more open to interpretation. For example, while fish

consumption safety is a basic requirement for a fishable water body; however, when nutrients are involved and when different species of fish have different sensitivities and requirements for nutrients in the water, the use and criterion definition become more problematic. In defining fishable it may be necessary to consider the species of fish or perhaps the recreational fishing success rate as criteria. Swimmable as a use (or aesthetics more generally) depends on user perceptions once the possibility of swimming related illness is removed. Perhaps the most difficult “use” to define and set criteria for is the most fundamental to the CWA – aquatic life support. With respect to this use, the basis for defining impairments under the CWA is its objective statement: “... to restore and maintain the chemical, physical, and biological integrity of the Nation’s waters.” Because the concept of “integrity” is not defined either conceptually or operationally within the CWA, mechanisms for defining impairments have changed over the years.

Virginia currently assesses a total of 5 designated uses for its freshwater resources: wildlife, aquatic life, swimming, fish consumption, and public water supply (Virginia DEQ 2004). Essentially all waterways are assessed for their suitability to support the first 4 of the 5 uses, and water bodies used for public water supplies are also assessed for their suitability to support that use.

One problem with the application of the designated use concept occurs when water bodies are intended to serve multiple uses. For example, maintaining water clarity levels in an impoundment that would be considered as suitable for contact recreation (i.e., a swimming designated use) may require maintenance of nutrient levels that are less than ideal for a fishing use (see Figure 4). Algal levels that are less than ideal for other uses may be found as suitable for waters used as municipal or industrial supplies. In such cases, it is necessary for the regulatory body to evaluate tradeoffs among the water-quality requirements of individual uses in establishing criteria.

### **Near-term Recommended Approach**

The committee’s recommended approach to freshwater nutrient criteria development is described below. Additional detail on the committee’s recommended approaches are included in Part II of this report.

#### EPA’s Guidance Criteria

The committee believes that the use of the 25th-percentile reference approach is inappropriate for Virginia. We are unaware of any factual basis to support this method’s fundamental underlying assumption: that 75% of all locations fail to satisfy the water-quality requirements of the Clean Water Act due to excess nutrients. Should these criteria be applied in Virginia, the committee believes that, as a direct result, DEQ would be required to conduct TMDL studies at numerous locations, regardless of whether the water bodies demonstrate evidence of designated-use impairment. This result would have a negative effect on Virginia’s environment by consuming DEQ’s resources that could otherwise be devoted to solving problems at locations where suitability for

designated use is impaired. There is a potential that EPA guidance criteria application could have a negative influence on Virginia taxpayers, as well, if DEQ resources were expended in a manner that does not achieve environmental protection.

Given the above, the committee recommends that DEQ endeavor to establish nutrient criteria by EPA deadlines (EPA requires that DEQ issue lakes and reservoirs' criteria in 2006, and streams and rivers' criteria in 2007), so as to avoid involuntary implementation of EPA guidance criteria.

### Designated Uses

Water quality standards incorporate criteria and use designations. The Clean Water Act and its implementing regulations state that criteria should be developed considering use designations. Given this CWA directive and the fundamental difference between nutrients and other pollutants that are more directly toxic, the committee recommends that DEQ should base its criteria development process upon the concept of designated use.

### Lakes and Reservoirs

All but two of the state's lakes are constructed impoundments (i.e., reservoirs). The committee recommends that, in the criteria development process, impoundments be considered separately from natural lakes, and that the bulk of Virginia DEQ's efforts in this area be devoted toward developing criteria for impoundments.

By definition, a constructed impoundment is an altered system. As such, impoundments can be considered as "unnatural" systems that require management. Given that a constructed impoundment will differ dramatically from both the free-flowing stream system that preceded its construction and any natural lake that may occur in similar topography, the committee believes that the logic for establishing criteria for impoundments on the basis of protecting designated use is especially strong.

Given that most impoundments are used and/or managed for recreational fishing and that species of recreational fish are generally at the upper trophic level, the committee believes that the status of recreational fish populations can be interpreted as an indicator of each impoundment's suitability for aquatic life. In the absence of fish population data that represent a number of impoundments and are comparable, the committee proposes the recreational fishery status, as rated by Virginia Department of Game and Inland Fisheries (VDGIF) biologists, be used as an indicator. Based on a preliminary analysis of VDGIF status indices' correspondence with nutrient levels, the committee believes that the state's impoundments should be classified for nutrient criteria development based on the types of fisheries that they support. While most of the state's impoundments support warm-water fisheries, the larger reservoirs also support cool-water fish species in their bottom waters, and some of the high-elevation impoundments are managed as trout fisheries. Each of these species types has different water-quality requirements.

We expect that the primary indicator of reservoirs' suitability for designated uses will be algal biomass, as represented by water-column chlorophyll *a* measurements, and we expect that nutrient criteria for lakes and reservoirs would be expressed as chlorophyll *a* concentrations. Water column nutrient concentrations (especially phosphorous, measured as total phosphorous or TP) can be expected to influence algal biomass levels. However, the committee is not prepared at this time to recommend whether or not criteria should also be expressed as TP concentrations, pending results of an analysis of the relationship between TP and chlorophyll *a* levels. Although it would be desirable from an ease-of-management standpoint to express criteria as water-column TP concentrations, we would not recommend such an expression unless analysis of water-quality data were to indicate a predictable correspondence between TP concentrations and chlorophyll *a*. Previous scientific studies demonstrate that impoundment morphometric features (such as retention time) can be expected to influence TP-chlorophyll *a* relationships, and therefore such features should also be considered as potential classifiers.

The committee recommends that DEQ should avoid specifying impoundment nutrient criteria for nitrogen. Although both nitrogen and phosphorous concentrations in the water column can be expected to influence algal biomass levels, phosphorous exerts a more direct control because some algal species (e.g., some blue-greens) are capable of utilizing atmospheric nitrogen and thus are not dependent on the water column for nitrogen nutrition. If criteria implementation and enforcement were to cause reductions of nitrogen inputs without a corresponding reduction of phosphorous, such actions could result in an increased risk of blue-green algal growth and consequent reduction of the water body's suitability for its designated uses.

The committee recommends that downstream loading effects should not be considered in establishing nutrient criteria for lakes and reservoirs. One reason for this recommendation is our expectation that algal-based criteria would require maintaining lower concentrations in the state's constructed impoundments than in associated flowing waters (the inflow and outflow rivers and streams). Therefore, if water quality conditions in receiving waters (such as the Chesapeake Bay) were found to justify consideration of downstream loading effects in criteria development, loading considerations could be applied more effectively in developing criteria for streams and rivers. Secondly, in-channel impoundments can help to reduce downstream loadings by trapping nutrients and by promoting denitrification. Therefore, downstream loading effects could be more appropriately applied in developing criteria for the outflow waters than to the impoundments themselves.

We expect that conflicts between the water quality requirements of various uses (for example: swimming vs. fishing) may be encountered in developing nutrient criteria, particularly for multiple-use impoundments. In addition, some of the state's impoundments were constructed as "best management practice" facilities for the purpose of controlling non-point source pollutants, including nutrients; in order for such impoundments to control downstream nutrient movement successfully, it may prove necessary to maintain water-column nutrient concentrations at levels above those suitable for designated uses such as contact recreation. For impoundments that are

servicing multiple uses, designation of appropriate criteria will require that tradeoffs among the requirements of those uses be considered and evaluated by DEQ.

We encourage DEQ to determine and consider the current uses for each of the state's impoundments in establishing criteria. For example, if the manager of an impoundment does not allow recreational swimming, the potential suitability of such an impoundment for swimming uses should not be considered in establishing criteria.

### Streams and Rivers

The AAC recognizes that establishing nutrient criteria for rivers and streams is a major challenge. Virginia's landscape spans a variety of terrains, ecoregions, and levels of development for human habitation. Although the AAC, as a group, has not yet come to a sufficient consensus to enable a detailed recommendation on how development of these criteria should be addressed, we do recommend that DEQ approach criteria development via a two-phased approach.

- Determine how measured algal biomass indicators correspond with the capability to serve the designated uses;
- Investigate the nature of relationships between water-column nutrients and algal biomass indicators, such as chlorophyll *a*.

Periphytic algae in wadeable streams and planktonic algae in non-wadeable streams should be considered as the primary indicator of use suitability.

We expect that algal impacts on the designated uses would be the primary focus of criteria development. However, the successful development of nutrient concentration-algal biomass predictive relationships applicable to Virginia streams and demonstrating a high level of statistical confidence would be beneficial to criteria development. Such relationships could enable specification of criteria as water-column nutrient concentrations, which can be measured for less cost than algal biomass indicators (especially periphytic algae) and can be controlled more directly by management actions.

EPA documentation states that downstream loadings should be considered in nutrient criteria development, and the committee recommends that DEQ consider these effects. A first step in the process could be a preliminary evaluation of the water-column nutrient levels in the tributaries of the Chesapeake Bay necessary for Virginia to achieve its Bay nutrient-reduction goals. Such an evaluation may be conducted by working with EPA and/or USGS to apply the Chesapeake Bay nutrient-loading model to Virginia tributaries. The results of this exercise can be assessed by referencing potential nutrient criteria values listed in Appendix B. If it is likely that water-column nutrient levels necessary to achieve a tributary system's loading reduction goals will be lower than the criteria that would be protective of local designated uses, the result could be an opportunity to conserve resources by excluding that tributary from nutrient criteria development activities that consider factors other than downstream loadings. However, because we understand the urgency of meeting EPA deadlines, we urge DEQ to conduct any such activities in an expedited manner so as to avoid delaying the criteria

development process and help ensure compliance with those deadlines. If the above studies prove to be more demanding, time consuming, and/or complex that the committee anticipates in making this recommendation, we would encourage DEQ to move forward with algal-based criteria, acknowledging that refinements and adjustments may prove necessary to accommodate downstream loading concerns over the longer term.

### Evaluation and Refinement

The committee believes that an effort to develop nutrient criteria that are appropriate for the state's waters will be a time- and resource-intensive process. The state's water resources reflect a variety of differences associated with variability of terrain, underlying geology, landscape development, canopy cover, morphometry, climate, and other factors. Because DEQ's water monitoring activities are conducted for the purpose of evaluating compliance with existing standards that do not include nutrient criteria, the resultant database is not ideally suited to the analysis of nutrient effects. Although we expect that DEQ personnel will devote both personal effort and professional expertise to the criteria development process, the inadequacy of the resources that DEQ is able to devote to criteria development, relative to the task's potential magnitude, is apparent. Thus, although we expect the criteria resulting from this process will be adequate for assessment of conditions that are common among the state's water resources, it is possible that such criteria would not accommodate or reflect the full range of variability represented by local conditions throughout the state.

Therefore, the committee recommends that DEQ build into nutrient criteria implementation a process for evaluating and refining the criteria. Such an evaluation and refinement process should have two components. Activities under the first component would occur in association with nutrient criteria violations. When violations occur, we would encourage a response that includes systematic determinations of (1) whether or not the water body's use designation is suitable for its actual use, and (2) whether or not the suitability for the designated use has in fact been impaired by nutrient over enrichment. We recognize that EPA will require that a TMDL study occur in response to any nutrient criteria violation. Therefore, we encourage DEQ to structure such TMDL's to include the above determinations and, if possible under EPA direction, to allow for avoidance of load allocation expenses in situations where actual water-body usage is found to have been not impaired. If the enforcement of the general criteria resulting from this process is found to produce systematic errors in impairment designations, responsible personnel should be assigned to both oversee the above determinations on a case-by-case basis and to recommend refinement of criteria elements, such as water-body classifications.

The above evaluation and refinement process, if implemented, should be sufficient to identify situations where nutrient criteria are unnecessarily restrictive. A corresponding process should be established in an effort to identify potential designated-use impairments that may not be indicated by criteria violations (i.e., where criteria are insufficiently restrictive). The Agency's monitoring programs would appear

to be an appropriate mechanism for such a process, although other mechanisms might also be pursued.

### **Longer-term Recommended Approach**

Over the longer term, we believe it would be advisable for DEQ to enhance its capability to consider differences among water bodies in applying nutrient criteria. For example, the committee has found that, as we attempt to find systematic relationships between nutrient levels and fishery status in the state's impoundments, effects of differences among these impoundments' characteristics become more apparent. Reservoir morphometric characteristics, stream gradients, stream canopy cover, and other physical factors can be expected to influence aquatic system responses to nutrient inputs and the consequent algal growth. We encourage DEQ to expand its database of information – both quantitative and narrative - that is descriptive of water-body use characteristics, and of water-body and monitoring-location features and physical conditions. Use designations and water-body classifications are two mechanisms that DEQ has available when evaluating water-body differences for nutrient criteria application. The evaluation and refinement process described above could be utilized by DEQ to advance its capability to apply these tools as needed to accommodate the vast range of water-body conditions that occur throughout the state, but an expanded descriptive database would be essential to that process. An improved capability by DEQ to consider differences among water bodies in applying nutrient criteria would create benefits for the Commonwealth. It would enable DEQ to protect the environment in a manner that both limits economic restrictions to only those water bodies unable to achieve clearly defined environmental goals and provides added assurance that such goals can be met.

Similarly, we believe that the Commonwealth would be well served by an enhancement of DEQ's capability to consider and quantify downstream loading effects in establishing nutrient criteria. Given the nutrient-reduction priorities associated with Virginia's receiving waters (including the Chesapeake Bay), Virginia's expanding population and economy, and the fact that non-point sources have become the primary water-borne nutrient sources, we expect that the challenges associated with water quality protection from nutrient effects are likely to increase in years to come. An increased capability to monitor and model non-point-source and point-source nutrient loading effects will enhance DEQ's capability to establish criteria that are appropriate for protection of coastal waters. Additionally, such capabilities would aid efforts to allocate tributary loadings in a manner that is consistent with both environmental priorities and continued economic progress. Also, they would provide opportunities to establish loading-allocation programs that are more effective than traditional approaches (which suffer from a lack of supporting information and therefore are uncertain in outcome), innovative, and (possibly) market based. While we recognize that resource limitations hinder major advances in this area in the short run, we suggest that DEQ adopt a strategic approach to encourage application of federal resources where possible. Such an approach may also be considered as a proactive response to the possibility that the

EPA might at some future time take a more aggressive approach to water-quality protection in the Chesapeake Bay, an action with the potential to cause negative economic impacts in Virginia if appropriate loading-allocation mechanisms are not available.

## Part II. Responses to DEQ Questions

### Approach

#### 1. Which, if any, of EPA's recommended approaches are appropriate, and why?

The committee believes that DEQ should emphasize application of scientific logic and analysis of Virginia water resource data in its criteria development process. The process should include the determination of the capability of water bodies to meet designated uses, as well as their biological, chemical, and physical status as required by the Clean Water Act's "integrity" clause. The committee believes that the establishment of nutrient criteria is an important activity for DEQ, as there is a strong potential for the criteria emerging from this process to have major impacts on the state's water resources and economy.

The EPA 304(a) guidance sets numerical criteria at points representing the lower 25<sup>th</sup> percentile of available data (either by ecoregion or from state-developed databases). The committee recognizes the simplicity of adopting the 304(a) approach for setting nutrient criteria but recommends that such a process not be considered by DEQ for application in Virginia:

- The primary problem with this approach is its inherent assumption that 75% of all locations should be defined as nutrient impaired. The committee is aware of no basis for this assumption in Virginia. If this approach were to be adopted in Virginia, the committee believes it would have a negative effect on water quality by diluting the available pool of TMDL and associated mitigation funding so that fewer resources are available to address those water resources that are experiencing severe problems. It is possible that such a result could have a negative effect on dischargers' regulatory compliance motivation, as well, if they were to believe that state and associated federal requirements were causing resource expenditures that were not having a significant impact by protecting water quality.
- If criteria resulting from the 403(a) process were to require nutrients below the levels desired to support beneficial activities (particularly fisheries), the result would include impairment to a designated use. (See Appendix C for additional information on the influence of nutrients on fish populations).

Another method recommended by EPA involves the use of reference conditions. Such an approach also has the advantage of simplicity in its application, but the committee also has strong reservations regarding its potential use in criteria development. As discussed in Part I, the concept of "least impacted" or "relatively undisturbed" references is not a reasonable means for assessing the majority of Virginia lakes, which are constructed impoundments. Because these impoundments were constructed for purposes associated with human uses such as recreation, flood control, and water supply and because the undisturbed status was a free-flowing stream, a reference condition approach is not reasonable.

The use of a reference condition approach may be more feasible for nutrients in streams and rivers than for lakes and reservoirs, but the committee would not favor EPA's recommended method for its application. First, most of the "relatively undisturbed" reference locations in Virginia can be expected to occur in forested areas, such as national forests, parks, and preserves. These areas differ in fundamental ways from the state's non-forested landscapes, which host residences, agricultural activities, industry, and transportation corridors. Water bodies are affected by watershed land use and land cover, as well as by the quality of any effluents they receive. These differences are especially relevant if biological indicators were to be used as a component of criteria development. For example, benthic macroinvertebrate communities will be affected directly by land use within the watershed and the riparian corridor because the land use affects the canopy cover and woody debris. Also, it can be expected that land cover will influence the watershed's capability to mitigate atmospheric deposition inputs. Given that nutrients differ fundamentally from other pollutants that are more acutely and directly toxic (Figure 3), we recommend that DEQ avoid setting criteria that use a direct application of the "relatively undisturbed" reference approach described by EPA documentation.

The committee believes that models developed from scientific studies in other locations may have application in development of nutrient criteria for Virginia, but such models should not be applied without a vigorous accompanying effort to determine whether or not the model results are consistent with the monitoring and measurement of Virginia's water resource conditions. As noted by Dodds et al. (2002), numerous models that define relationships between water column nutrient status and algal biomass are available, but the results from the use of these models can differ substantially based on the model selected for a given situation.

Given that well over 50% of the state drains into two nutrient-sensitive Atlantic estuaries, Chesapeake Bay and Albemarle Sound, impact on downstream water bodies should be a fundamental component of nutrient criteria development.

An appropriate basis for developing nutrient criteria for the state's constructed impoundments consists of an assessment of the water bodies' suitability to meet its designated uses. This statement is based on the following observations:

- All but two of Virginia's lakes are constructed impoundments. Therefore, the concept of an "undisturbed reference" is not a reasonable means for assessing constructed impoundments, since the undisturbed status of such sites was a free-flowing stream. Virginia's two natural lakes are located in fairly unique physiographic conditions. These lakes cannot serve as a "relatively undisturbed reference" for the variety of physiographic conditions that host the state's constructed impoundments.
- One purpose for constructing some of the smaller impoundments was for pollutant retention, and the larger reservoirs built in river channels perform this function. This function is especially important given the nutrient-sensitive status of coastal receiving waters (e.g., Chesapeake Bay, Albemarle Sound). EPA documentation is specific in stating that states may consider the impoundments' nutrient trapping functions when developing nutrient criteria (U.S. EPA 2000g,

Chapter 7).

A recommended approach to data analysis for use in nutrient criteria development for the state's constructed impoundments is included as Appendix A to this report. The committee has not yet reached a consensus regarding how nutrient criteria development for rivers and streams should be approached.

## **2. Should Virginia consider effect-based criteria derived by finding correlations between nutrient enrichment and negative changes in biological variables?**

The committee believes that the development of effect-based criteria is a desirable approach and that relationships between nutrient enrichment and negative changes in biological variables do, in fact, exist. A logical approach to the task of investigating these relationships is to break it down to two components:

- Determine how measured algal biomass indicators correspond with factors fundamental to criteria development: capability to serve the designated uses;
- Investigate relationships between water-column nutrients and algal biomass indicators, such as chlorophyll *a*.

The remainder of this response will address the first component above. Given the wealth of scientific knowledge about the role of nutrients in biological systems, it is reasonable to expect that a well-executed scientific study should be able to detect such relationships. To build adequate models requires that we have sufficient variation of biological and nutrient data. Given that Virginia DEQ's monitoring programs were developed for purposes that do not include nutrient criteria development, we have serious doubts whether existing monitoring data will prove be sufficiently robust to derive such criteria on an ecoregion, or even a statewide basis, given the following factors:

- It is not clear that DEQ's monitoring data represent adequately the range of nutrient conditions that will be required to define nutrient-algal biomass indicator (chlorophyll *a*) models. For example, if we have dominantly low values of P and only a few high values (as is the case with the lakes' ambient monitoring database), it is not likely that we can find adequate relationships based on higher phosphorous concentrations. Data developed from probabilistic sampling of streams is unlikely to produce adequate regression relationships unless there are a reasonably high number of poor (overly enriched) sites.
- Existing data may lead to weak relationships due to the presence of covariates. For example, in order to relate chlorophyll *a* and P, we would need to have data that exhibit variation in both chlorophyll *a* and P. If chlorophyll *a* covaries with N, then this would confound the relationship of P with chlorophyll *a*.
- Existing data may not include all factors necessary to understand relationships between water column nutrients and algal biomass indicators. For example, several scientific studies have found that factors such as water velocity and time since the most recent storm-flow event will influence nutrient-algal relationships

in rivers and streams, but DEQ's ambient monitoring data do not contain variables that represent these factors.

- The bulk of scientific literature addressing nutrient-algal relationships in flowing waters focuses on periphytic algae, not water column chlorophyll *a*. To our knowledge, the Virginia DEQ ambient monitoring program does not routinely measure or monitor periphytic algae.
- The DEQ probabilistic monitoring activities occur in spring and fall, and obtain only one measurement of water-column nutrients at each site. A single observation of water-column nutrients is not adequate to characterize variability, and the probabilistic monitoring data is not available to represent summer conditions when nutrient impairments can be expected to be most severe.

### Lakes and Reservoirs

A proposed approach for analyzing DEQ data, for the purpose of assessing nutrient-algal indicator relationships, is reviewed in Appendix A of this report.

### Rivers and Streams

Fundamental to the development of criteria for rivers and streams should be an investigation of the correspondence between in-stream nutrients and algal biomass, quantified either as a direct measurement or as an indicator such as chlorophyll *a*.

Results of research conducted in other areas have been highly variable but, in general, these researchers have found relationships between nutrients and benthic chlorophyll in streams to be weaker than the corresponding relationships in lakes. For lakes, nutrient levels are commonly found to be responsible for greater than 50%, and sometimes for as much as 60% to 70%, of the variation of water-column chlorophyll *a*. When similar studies are conducted in rivers and streams, nutrient-chlorophyll *a* coefficients of variation are generally much lower, in large part because numerous other factors – both physical and biological – affect benthic algae. Variables such as flood frequency and stream velocity represent the tendency of hydraulic disturbances to scour the stream bottom and remove periphytic algae. Populations of macroinvertebrate grazers can have similar effects. Light (as affected by canopy cover) and temperature also affect the algal response to nutrient levels. As a result, many studies of nutrients and periphyton in freshwater streams have not found strong relationships.

For example, Bourassa and Cattaneo (1998) conducted a study of the relationship between periphyton biomass and potential controlling factors in 2nd and 3rd order streams in Laurentian, Quebec. They found that periphyton biomass was not significantly related to nutrient concentrations. Of the variables measured, they found that the strongest relationships to algal biomass were exhibited by stream depth and velocity at sampling points. The investigators found grazer biomass to be positively correlated with TP and believed grazers to be responsible for the lack of a stronger TP-chlorophyll relationship. Working in eight Ottawa streams selected to represent a strong trophic gradient, Cattaneo et al. (1997) found that, while TP differences explained 25% of the periphyton biomass variation, differences in suspended seston explained 40%,

and a model that combined measured seston and an indicator of substrate size explained 44%. Biggs (2000) conducted a similar study, working in 25 New Zealand streams. He found “days of accrual” (days since a storm flow totaling >3 times the median streamflow) to be the strongest explanatory variable, accounting for 39% of the variation in mean, and 62% of maximum, monthly benthic algal chlorophyll. Working at 89 randomly selected mid-Atlantic stream locations, Pan et al. (1999) found only a weak relationship between benthic chlorophyll *a* and TP ( $r = 0.29$ ,  $r^2 = 9\%$ ). Dodds et al. (1997, 2002) conducted longitudinal studies of factors influencing benthic algal biomass in streams using databases derived from their own research, and from scientific literature describing studies conducted in temperate climates worldwide including North America, Europe, and New Zealand. In these studies, the “best” models of TP-benthic algal chlorophyll relationships were accompanied by  $r^2$  values ranging from 8% to 32%, while  $r^2$  values for the corresponding TN models ranged from 20% to 37%, and models combining TN and TP were able to explain from 35% to 43 % of the variation in benthic algal chlorophyll *a*. Dodds et al. (2002) found that a number of other factors demonstrated statistically significant relationships with seasonal mean and/or maximum chlorophyll *a*, including stream gradient (-), temperature (+), substrate type (-), and maximum discharge (-).

On the other hand, some studies have found very strong relationships between in-stream nutrients and periphytic chlorophyll or biomass. For example, Van Nieuwenhuysse and Jones (1996) compiled a database from 292 North American locations. For each location, they compiled mean May-September values for TP and chlorophyll *a*. The resultant relationship was curvilinear (with steepness declining at increasing TP concentrations), with an  $r^2$  of 67%. Working at 22 sites in the Ozarks, Lohman et al. (1992) found that nutrient levels explained 47% to 60% of periphyton biomass variability. Working in 13 Canadian rivers, Chetelat et al. (1999) found that TP levels explained 56% of algal biomass variation.

Given the wide variability of results obtained by researchers working in other areas, the committee believes strongly that the only way to develop realistic nutrient criteria for Virginia streams is through applied and targeted investigations conducted in Virginia streams. The committee believes that, for the vast majority of Virginia streams, such a study would need to quantify periphytic algae, because water column chlorophyll measurements can be expected to correspond only weakly (if at all) with total photosynthetic activity.

One of the reasons why results of previous research have been so variable is that streams vary in their response to nutrient enrichment depending on their size, location, basin characteristics, and other features. Stream nutrient monitoring programs need to be tailored to meet specific stream types and situations—one size does not fit all. For example, low-order woodland streams are covered by streamside vegetation and receive very little light and thus generally support low algal growth. Energetics (food sources) in forest stream types is likely to be dominated by litter (e.g., leaves, sticks, floral parts) from streamside vegetation. Most such streams probably receive limited excess nutrients from the landscape. However, some such streams can have moderate growths of shade-tolerant algae like diatoms. Thus monitoring algal biomass (mass accumulated by growth through time) as chlorophyll *a* and ash free dry mass (AFDM)

on either natural or artificial streambed materials (substrates) may prove fruitful in detecting the effects of excess nutrients. Because high flows may scour algal biomass from substrate surfaces, frequent monitoring is required to prevent missing algal responses.

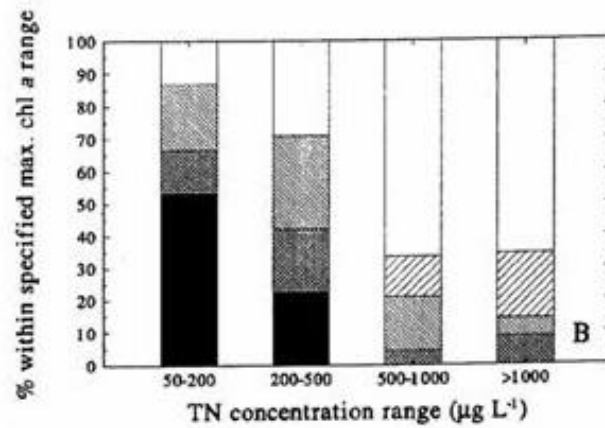
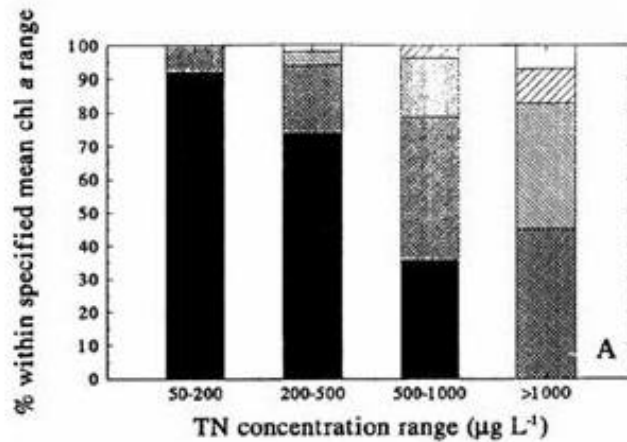
Mid-order streams draining forests may receive sufficient light to support algal growth such that monitoring chlorophyll *a* and AFDM on substrates should be an appropriate way to estimate the nutrient condition. Combining the measures of algal biomass with the ratio of scrapers (invertebrates like snails and certain aquatic insects that scrape algae and other biofilm constituents from surfaces) to shredders (invertebrates that consume microbially “conditioned” leaf litter) may increase information regarding nutrient effects. For example, a Scraper/Shredder greater than 1.0 would indicate that photosynthetic algae are important contributors to the energy flow in the stream.

In open, shallow streams where light is plentiful, algal biomass should be an appropriate indicator of nutrient loading so measuring chlorophyll *a* and AFDM on natural or artificial substrates should suffice. In deep (i.e., deeper than waist-high), slowly moving streams (such as the lower James), most algae may be present in the water column as plankton (floating microorganisms). Measuring chlorophyll *a* in water samples for these deeper waters may be more indicative of the nutrient condition than measuring algae biomass on solid substrates.

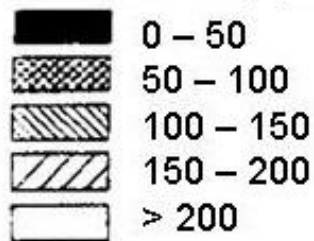
Dodds et al. (1997) applied a technique that may prove useful to the analysis of data on algal biomass in Virginia streams, if similar data for Virginia streams could be obtained. Their goal was to define nutrient concentrations that correspond with nuisance levels of periphytic biomass (100 mg/m<sup>2</sup> mean, 150 mg/m<sup>2</sup> maximum). They approached this task by applying 3 different methods independently:

- Application of a nutrient-periphyton model derived from scientific literature.
- Application of a probabilistic technique derived from Heiskary and Walker (1988), which determined the frequency at which critical chlorophyll *a* levels are exceeded when water column nutrients are within a specified range, using in-stream data from multiple locations (see Figure 5).
- Identifying a number of locations at which periphytic algal levels were considered to be at an acceptable level, and determining the associated water column nutrient levels.

These authors report that the nutrient levels derived from all three techniques converged, which enabled them to define in-stream nutrient criteria that were protective of acceptable water quality.



**Chl-a levels (mg/m<sup>2</sup>):**



**Figure 5.** Probability distribution of mean and maximum chlorophyll *a* ranges from locations sampled in the Clark Fork system of western Montana (Dodds et al. 1997). Multiple observations were obtained at each location. A research goal was to define target nutrient concentrations that, when not exceeded, will “generally yield acceptable levels of chlorophyll *a*” (<100 mg/m<sup>2</sup> mean; <150 mg/m<sup>2</sup> maximum). From this analysis, the researchers concluded that maintaining TN levels between 200 and 500 µg/L would yield acceptable levels “in most cases.” By combining these data with the results of two other analyses, the researchers concluded that 350 µg/L TN could be adopted as a “provisional target level that would allow some external input of TN to the Clark Fork system and yet should avoid frequent episodes of excessive algal growth.”

### **3. Should criteria development be tied to ecological endpoints indicating impairment?**

Conceptually, the committee sees a link between criteria and ecological endpoints that indicate impairments as critical to the long-term acceptance and defensibility of developed criteria. Ideally, criteria can be firmly linked to carefully selected ecological endpoints. Not only does this put the criteria on a clear cause-effect basis, but it also makes a transparent link to the underlying reasons for the criteria selection. The cause-effect relationship between nutrients and many biological response variables have been well studied.

Operationally, however, the situation is not quite so straightforward, as explained below.

#### Lakes and Reservoirs

Impairment designations will be intended to identify locations where nutrient enrichment is having a negative effect on the suitability of a water body for aquatic life. In Virginia's constructed impoundments, the committee believes that the status of the recreational fishery can be considered as an indicator of the impoundments' suitability for aquatic life. Given that species of recreational fish are generally at the upper trophic level, the health of recreational fish populations can be interpreted as an indicator of ecosystem health as well as suitability for the aquatic life designated use.

A practical problem faced in criteria development is how to assess nutrient effects on recreational fish populations. The committee's preferred method for approaching this problem would be to assess fish populations in a sample of the state's impoundments selected to represent the full range of nutrient concentrations occurring within the state's reservoirs. Ideally, the full range of nutrient concentrations would be represented by the reservoir sample within each ecoregion, so as to enable investigation of ecoregion effects. Obstacles to implementation of such an approach include the following:

- Preliminary analysis of DEQ monitoring data indicate that monitored reservoir nutrient concentrations are unevenly distributed, with only a small number of impoundments representing the high end of the nutrient concentration range.
- Lakes with high nutrient concentrations tend to be clustered in eastern Virginia, thus confounding nutrient concentration with possible ecoregion effects.
- The resources (e.g., finances, labor, equipment) required to assess fish populations in a sample of Virginia lakes using a consistent and adequate methodology are not apparent.
- Morphometric features, such as retention time, can also be expected to influence algal response to nutrient inputs.
- Conduct of such a study would be very costly and necessary resources are not readily available.

Given the above problems, the committee has proposed an alternate methodology that is described in Appendix A of this document.

### Rivers and Streams

The AAC believes that the most defensible approach would be to define in-stream benthic algal levels that correspond with impairment. Nutrient criteria would be comprised of those algal levels and – if defensible and scientifically verified relationships between benthic algal levels and in-stream nutrient concentrations are found to exist – the associated in-stream nutrient concentrations.

However, at this point we do not see a clear process for identifying algal levels that would correspond with impairment. As noted earlier (Part I; Part II, Question 1), we do not see the logic of applying an arbitrary percentile to the distribution of Virginia values. We do not see nutrient concentrations of “relatively undisturbed reference” locations, such as national forest streams, as a logical means for defining criteria that would be applied in developed areas of the state. Although both of the above methods would be capable of defining clear distinctions, those distinctions would not be based on the evidence of impairment in relation to the designated uses.

Another potential approach could be to seek relationships between benthic algae levels and benthic macroinvertebrate community indices. The logic for application of this method would be that the benthic macroinvertebrate community can be expected to respond negatively to nutrient over enrichment and, because Virginia DEQ utilizes benthic macroinvertebrates as bioindicators, ample data on benthic macroinvertebrate status in streams throughout the state are available. However, the committee believes that, because nutrient effects on benthic macroinvertebrates tend to be far more subtle than effects exerted by other stressors, including some which are both widespread and inadequately monitored by DEQ’s ambient monitoring protocol (e.g., sediments), such an approach would be unlikely to yield success.

Although the committee has not reached a consensus regarding how to approach nutrient criteria development for rivers and streams, we have discussed several methods including a modified reference approach that would consider land-use effects and designated use suitability, scientific literature studies, and an approach similar to that described by Dodds et al. (1998) and described in response to Question 2.

### Form

**4. Are the 1987 TAC water body type, parameter, and concentration recommendations for the nutrient enriched waters regulation currently applicable, including the TAC recommendation that nitrogen was not an appropriate criterion?**

The committee sees the 1987 TAC report as having continued relevance to the current process. The recommendation to consider criteria on a water body-type basis is also attractive. However, the conclusions of the 1987 TAC should not be given *a priori*

weight greater than other sources to be considered in criteria development. It should be noted that discussion of the relationship between ecological endpoints and numeric criteria did not dominate the 1987 committee.

## **5. What are the most likely metrics for streams, lakes, and estuaries?**

The committee sees this as a critical question. The answer to this question should be based on the analysis of data defining relationships among measured nutrients, ecological response variables, and water body conditions that are protected by the Clean Water Act. In the absence of data and analyses describing the relationships between these factors, the committee is not prepared to answer this question.

## **6. Should the criteria be causal variables (nitrogen and phosphorus concentrations); or be response variables like water clarity, chlorophyll *a*, trophic state indices (TSIs), or other algal indices; or both?**

In the absence of data and analyses describing the relationships between these factors, the committee does not have a clear recommendation on how to approach this question at this time. Clearly, nutrients are the cause of nutrient impairments. Therefore, it would be preferable to identify nutrient levels that correspond with impairment if causal links can be identified with a sufficient level of certainty. However, given the complexity of ecological processes governing nutrient impairment, the committee wishes to reserve its opinion on how this question should be approached until data analyses have been completed and results are available.

An advantage to defining criteria as water-column nutrient levels would be that control actions intended to alleviate criteria violations could be prescribed with greater certainty of success. If response variables are selected for criteria, then the ability to predict the consequences of any pollutant or pollution control actions, relative to the water body's impairment status, will be subject to more significant prediction uncertainty than if the criteria are based on causal variables. Hence there is a tradeoff between making surrogates close proxies for use (response variables) and the ability to predict (with modest error bands) the effects of control actions on the surrogate criteria. If the water quality management process recognizes and accommodates such uncertainty in implementing control programs, then it will be most desirable for the criteria to be response variables.

An advantage to defining criteria as response variables is that such criteria can be linked more directly to ecological endpoints than can nutrient concentrations. We expect that DEQ's capability to establish tight relationships between response variables and ecological endpoints would be improved if more advanced response-variable monitoring methods were applied. For example, data defining algal species distributions, in addition to chlorophyll *a* measurements, could allow development of an ecological endpoint indicator with greater sensitivity than measuring chlorophyll *a* alone. Given that such advanced indicators would be more difficult and expensive to measure than chemical water quality, however, it may be worthwhile to explore a decision structure such as that which follows:

- If N and/or P are below certain levels (let's call them  $N_{ref}$  and  $P_{ref}$ ), the water body is not impaired.
- If N and/or P are above certain levels (let's call them  $N_{imp}$  and  $P_{imp}$ ), the water body is impaired.
- If  $N_{ref} < N < N_{imp}$  and/or  $P_{ref} < P < P_{imp}$ , then measure advanced response variables to determine whether or not the water body is impaired.

By adopting the above or comparable approach, DEQ would be able to focus greater resources on measuring ecological response variables at each suspect location than would be possible if such measurements were to be performed at all monitoring locations, regardless of nutrient status. Such an approach would be most useful if criteria were expressed solely as response variables that are costly or difficult to measure, such as periphytic biomass.

## **7. What approaches should Virginia take to demonstrate where nitrogen criteria are not needed for freshwater lakes and reservoirs and streams and rivers?**

The AAC sees the question of how to address nitrogen in freshwater nutrient criteria as critical and complex. Conventional wisdom – that the vast majority of freshwater systems are P limited – leads to the “easy” answer (“nitrogen doesn’t matter ...”), but the easy answer is not necessarily correct. Therefore, we have focused some effort on investigating the role of N in lake and reservoir systems for the current year.

### Lakes and Reservoirs

#### *Scientific Background:*

##### *1. Phosphorous limitations should be the primary focus for nutrient criteria development.*

In lakes, algal community populations are commonly considered to be limited by the availability of phosphorous. Numerous studies have demonstrated that algal densities are strongly influenced by P concentration. For example, in a study of 19 northern lakes, Dillon and Rigler (1974) demonstrated a strong linear relationship between water column TP concentration at spring overturn and summer chlorophyll *a* concentrations ( $r \sim 0.9$ ). Rast et al. (1983) summarized studies that demonstrate consistent relationships between annual areal phosphorous loading and three nutrient related response variables (mean summer chlorophyll *a*, Secchi depth (SD), and hypolimnetic oxygen depletion rate) for selected U.S. lakes and reservoirs. They observed decreasing chlorophyll levels and increasing Secchi depths in 10 lakes that experienced P loading declines. Working with a data set of approximately 75 lakes from North America and Europe, Schindler (1978) found that “A high proportion of the variance in both annual phytoplankton production and mean annual chlorophyll could be explained by annual phosphorous input (loading), once a simple correction for water renewal was applied.” Although we have cited some of the older, classic studies here, the fact that phosphorous concentrations exert a major influence over algal populations in lakes and reservoirs remains well established in the current scientific literature. As noted by

Schindler (1978), the fact that some algal species are capable of obtaining N (but not P) nutrition from atmospheric sources dictates that P supplies will exert the primary control on primary productivity in most lakes.

*2. When limited or co-limited by N, algal communities can be expected to respond to changes in available N supplies.*

In any ecosystem at any given time, photosynthesis will be limited by the necessary abiotic factor that is present in the least quantity relative to requirements. Although algal communities in lakes are generally considered to be P-limited, conditions also occur where algal growth is limited by a lack of N.

Because some algal species have a capability to fix atmospheric N, situations where freshwater algae are limited solely by N are not common, but N limitation can occur. For example, Morris and Lewis (1986) found that water-column soluble inorganic N levels in three of the eight Colorado lakes that they studied declined to levels indicating N limitation during the mid-summer months. Several studies have found that short-term N limitations can occur commonly in systems where seasonal means give no indication of N limitation (Barica 1990, Matthews et al. 2002). "Patchy" distributions of algal species and nutrients in aquatic systems, especially when stratified, can cause N limitations to occur within microenvironments even when the system average nutrient concentrations do not indicate such condition (Hyenstrand et al. 1998). Long-term N limitation is rare because some algal species are capable of fixing atmospheric N, although it can occur in systems with conditions, such as micronutrient deficiencies, that inhibit the growth of N-fixing algae.

A more common situation is where algal communities are co-limited by both N and P. Co-limitation occurs because numerous species are present and because algal community and species vary in the proportions in which they require N and P. At a given N/P ratio in the co-limitation range, some of the species present may be limited by N and others by P (Suttle and Harrison 1988, Dodds et al. 1989). When algal populations are co-limited by N and P, populations can be expected to respond to changes in the supply of either nutrient.

A number of researchers have found that co-limitation of primary productivity by N and P is common in lakes. As reported by Dodds et al. (1989), "statements that phosphorous is the major nutrient controlling primary productivity in freshwater systems ... should not be taken to mean that phosphorous is the only nutrient limiting productivity in all systems." An example of co-limitation is presented by Dodds et al. (1989). These researchers fertilized algal cultures withdrawn from a Montana reservoir with  $\text{NH}_4^+$  and  $\text{PO}_4^{3-}$  in proportions equivalent to the Redfield Ratio, which represents the typical or average proportion of N to P in algal biomass tissue, and with equivalent amounts of  $\text{NH}_4^+$  and  $\text{PO}_4^{3-}$  alone. The  $\text{NH}_4^+$  addition alone stimulated production by 22%; the  $\text{PO}_4^{3-}$  addition increased production by 18%; and the combined addition boosted production by 40%. In reviewing published studies of whole-lake fertilization experiments, Elser et al. (1990) found that enrichment by N and P, in combination, was often required to enhance algal growth, and conclude that their results provide little support for the conventional wisdom that lake communities are almost always limited

solely by P. A number of studies analyzing data from multiple lakes have found that regressions using both TN and TP can explain more variance in epilimnetic algae (or algal indicators such as chlorophyll) than can regressions using TP alone (Smith 1982, Canfield 1983, McCauley et al. 1989, Prairie et al. 1989). This result occurs in part because N-fixing species devote energy to N fixation and are thus less productive in response to P and sunlight inputs than non-N-fixing species (Smith 1983, Howarth et al. 1988, Suttle and Harrison 1988). It may similarly be reasoned that species using oxidized compounds such as nitrite and nitrate for an N source must devote energy to a chemical reduction to produce ammonium, and it is likely that such systems would also exhibit a lower productivity response for a given concentration of N. In a study of southeastern lakes and reservoirs (using a data set comprised primarily of constructed reservoirs), Reckhow (1988) found that both TN and TP were positively but weakly correlated with chlorophyll *a* ( $r = 0.449$  and  $0.338$ , respectively).

### *3. Changes in blue-green populations can be expected to occur in response to manipulation of lake nutrient levels.*

The primary nitrogen fixers in most algal systems are blue-green algae, also known as cyanobacteria. Blue-green algae can be expected to have a negative effect on the capabilities of lakes to serve uses such as aquatic life support, recreation, and water supply. Cyanobacteria are less suitable as food sources for zooplankton than other phytoplankton species; therefore, such blooms of blue-green algae will have a negative effect on higher trophic levels, including fish. Some cyanobacteria species release toxins to the water column that can be harmful to consumer organisms, including zooplankton and fish. Blooms of some cyanobacterial species will also cause water clarity to exhibit greater decline than occurs in response to an equivalent biomass of green algae species, this having a negative effect on the recreational suitability of water bodies. In reservoirs used as water supplies, cyanobacterial blooms can result in increased treatment requirements for several reasons. Bloom conditions of cyanobacter species have been observed to (1) reduce filter operation efficiency due to the presence of floating mats, (2) increase intensity and frequency of taste and odor episodes due to the secretion of extracellular metabolites (ECM's), and (3) enhance the formation of regulated disinfection by-products from the reaction of chlorine with ECM's. Cyanobacteria species vary widely in growth habits and characteristics, and not all cyanobacteria are N fixers (Hyenstrand et al. 1998, Dokulil and Teubner 2000).

It has long been known that changes of lake nutrient levels can cause changes in cyanobacterial populations, but the mechanisms by which changing nutrient levels affect blue-green algae populations remain poorly understood.

The "resource ratio" theory of algal community development is based upon the observation that the composition of most living cells includes N and P in relatively constant proportions. Regarding the effect of nutrient levels on algal communities, resource ratio adherents believe that, "the optimal N:P ratio for a given species is equal to the ratio of its minimum cell requirements for these elements," and thus relatively low N/P ratios in the epilimnion will favor a shift of algal community structure to N-fixing blue greens (Bulgakov and Levich 1999). The "Redfield Ratio" (16 atoms of N to each P atom, or about 7g N to 1 g P) is commonly cited as being optimal for algal communities,

but in fact the cellular composition of algae varies widely by species, ranging from 7:1 to 45:1 by atoms (Suttle and Harrison 1988), equivalent to an approximate range of 3:1 to 20:1 by weight. Schindler's (1977) study of nutrient enrichment effects in Canadian experimental lakes is often cited as demonstrating that cyanobacterial blooms can occur in response to decreasing N/P ratios. Schindler's fertilization experiments caused algal blooms in all cases. Fertilization with low (~5 by weight) N/P ratios by mass caused the resultant blooms to be dominated by blue-greens, while fertilization with higher N/P ratios (~14) resulted in green algae dominance. Schindler noted that the TN/TP ratio in the low-N/P fertilized lakes remained at ~14, despite the lower nutrient ratio of the fertilizer additions, as the algal community shifted toward N fixation and fixation rates increased.

In a study of 17 lakes from around the world, Smith (1982) found that blue-green algae tend to be rare when growing season N:P ratios (by weight) exceeded 29:1, while the proportion of blue-green algae to total populations was more variable at lower ratios. In a later study using similar methods, Smith found that blue-greens' domination was also rare when N/P ratios exceeded 22:1 (Smith et al. 1995, Smith and Bennett 1999). In 1986, Smith analyzed data from 20 Alberta lakes and concluded that the blue-green algae respond to both light levels and TN/TP ratios. Smith concluded from this study that "minimizing TP loading (to minimize chlorophyll-related light attenuation) and managing loading N/P ratios to obtain high TN/TP ratios in the lake may both be necessary if one's objective is to minimize the relative biomass of blue-green algae." Smith and Bennett (1999) reviewed data generated by numerous researchers that they interpreted as supporting resource ratios as mechanistic determinants of algal community composition, stating "strong agreement ... that resource-ratio theory provides a very plausible explanation" for differences in phytoplankton community structure among lakes. Strict proponents of the resource-ratio theory believe that N fertilization (that increases the N/P ratio beyond critical levels) can remedy blue-green algal blooms in situations where P reduction is difficult (Levich 1996). Lathrop (1988) reports that an effort to remedy cyanobacterial blooms in a Wisconsin lake by applying this technique was unsuccessful.

Other researchers disagree that resource ratios are mechanistic determinants of algal community composition but agree that N/P ratios have influence. In 1987, Trimbee and Prepas (1987) published a study that disputed the conclusions drawn by Smith (1986). These investigators reanalyzed Smith's Alberta data, adjusting the analysis for data problems that they believed to cause bias. Their investigations concluded that a model predicting relative blue-green algae (i.e., the proportion of total algal population consisting of blue greens) as a function of TP explains a higher proportion of the variance than prediction as a function of N/P. Watson et al. (1997) synthesized seasonal mean algal biomass, algal species composition, and total P data for 204 north temperate lakes. They categorized algal species into 6 major taxonomic groups and analyzed the relationship of each species group to TP independently. They found that, at very high and very low TP levels, one or a few groups tend to dominate the community while, at intermediate TP levels (~10-30  $\mu\text{g/L}$ ), populations were more evenly distributed. They also found that as TP increases beyond ~30  $\mu\text{g/L}$ , non-edible taxonomic groups (diatoms and blue greens) become increasingly dominant. Of the six

functional groups, cyanobacteria populations show the strongest relationship to TP, being nearly absent at  $<10 \mu\text{g/L}$  but the predominant algal form ( $>50\%$  of total biomass) at  $> 60 \mu\text{g/L}$ . This same group of authors conducted a later analysis of 269 observations of algal biomass, community composition, and nutrient levels from 69 lakes around the world (Downing et al. 2001), and concluded that “the risk of water quality degradation by cyanobacteria blooms is more strongly correlated with variation in total P, total N, or standing algal biomass than the ratio of N:P.” These authors state that “the risk of cyanobacteria dominance is only 0–10% between 0 and  $30 \mu\text{g/L}$  of total P, rising abruptly to about 40% between  $30$  and  $70 \mu\text{g/L}$ , then asymptoting at around 80% near  $100 \mu\text{g/L}$ ,” and that risk levels rise to  $>10\%$  when chlorophyll *a* concentrations are  $>10 \mu\text{g/L}$ , which generally signified transparency  $<1$  m. Dokulil and Teubner (2000) state that “the relative probability for prolonged dominance by cyanobacteria is significantly reduced at phosphorous concentrations  $< 100 \mu\text{g/L}$ ...” but “the absence of cyanobacteria is better guaranteed at levels below  $50 \mu\text{g/L}$ , although dominance is still possible.”

Reynolds (1999) provides a scientific explanation for the total-resource arguments, stating that resource-ratio models “do not predict the outcome when growth rates become saturated.” Reynolds states that algal species differ in the rate at which they are able to take up nutrients as specific chemical forms and at given concentrations, and that these differences help define community structure. He supports these observations by citing the results of laboratory culture experiments that monitored algal growth and species composition under controlled soluble nutrient levels.

Reynolds (1999) notes that, although he believes that scientific evidence does not support a mechanistic role for resource ratios, N/P ratios do play an indirect role in determining algal community composition. As non-N-fixing algae proliferate at a low N/P ratio, for example, water-column dissolved N is likely to become depleted before dissolved P, thus creating conditions favorable for a shift of community composition toward N fixing species.

Several characteristics of blue-green algae contribute to their capability to dominate under eutrophic conditions (Dokulil and Teubner 2000). Cyanobacteria are known to have lower light intensity requirements than other algal species, which allows them to have a competitive advantage under darkened environments such as those that occur under eutrophic conditions. Non-N-fixing cyanobacteria are believed to have a limited capability to assimilate N as  $\text{NO}_3^-$  but are highly competitive for ammonium N (Blomqvist et al. 1994, Hyenstrand et al. 1998). This N species preference can provide a competitive advantage for non-N-fixing cyanobacteria under eutrophic conditions that accelerate the accumulation of particulate organic N and consequently lead to  $\text{O}_2$  depletion in the subsurface. The resultant anoxic conditions allow conversion of organic N to  $\text{NH}_4^+$  but hinder  $\text{NH}_4^+$  conversion to  $\text{NO}_3^-$  and stimulate denitrification losses of  $\text{NO}_3^-$  to gaseous forms. Cyanobacteria tend to be excellent competitors for P at relatively high concentrations characteristic of eutrophic systems but less successful at lower concentrations (Suttle and Harrison 1988). Some species of cyanobacteria are able to regulate their buoyancy, allowing them to move vertically in the water column to take advantage of differential vertical availability of light and nutrients (Klemer and Kanopka 1989), such as those that occur in eutrophic systems. Some of these

buoyancy-regulating species also have a capability to assimilate and store P internally, allowing them to obtain P from the sediments and gain a competitive advantage under conditions of P limitation (Hyenstrand et al. 1998).

Our review of the scientific literature makes it clear that numerous factors influence cyanobacterial dominance, and that – in part due to the variety of species and species properties within the cyanobacteria group – the scientific capability to predict conditions that will cause cyanobacterial blooms is still fairly crude. There is no single factor or theory that adequately explains or predicts cyanobacterial dominance (Hyenstrand et al. 1998). For example, although cyanobacterial dominance often occurs under eutrophic or hypertrophic conditions, there have been instances where cyanobacterial dominance has occurred under oligotrophic conditions. Working in shallow Danish lakes, Jensen et al. (1994) found different groups of cyanobacteria to be dominant under low N/P and high P conditions.

It has also been observed that inorganic N speciation can affect algal species dominance. One study of the Occoquan Reservoir found that changes in the inorganic nitrogen supply from ammonium to nitrate were accompanied by shifts in algal species dominance away from the cyanobacter and towards green algae and diatoms (T. Grizzard, personal communication). The observed shift was also found (at least anecdotally) to have beneficial impacts on water treatment operations.

In addition to nutrient-related factors, other water body properties can contribute to cyanobacterial success. Because cyanobacteria have a high affinity for carbon as  $\text{HCO}_3^-$ , conditions of low pH / high alkalinity have been demonstrated to increase the potential for cyanobacterial dominance. Elevated water temperatures and high availabilities of trace elements are also favorable conditions for cyanobacterial development. Because cyanobacteria have higher requirements for trace elements than other algal forms, their development is hindered in water bodies with low trace element concentrations.

We interpret the above information to indicate that a logical strategy for reducing potentials for cyanobacterial blooms through nutrient criteria implementation should include minimizing TP loading, avoidance of management strategies that create or maintain N/P ratios in the N limitation range, and avoidance of strategies that cause N/P ratios to decline when lakes are N and P co-limited. The AAC recognizes that such a strategy may, at times, create conflict between criteria developed for local water quality management and those developed as part of a management strategy for downstream estuarine systems, such as the Chesapeake Bay. For example, some approaches to maintaining sufficiently high freshwater N:P ratios to safeguard against cyanobacterial blooms, may result in the export of excess nitrogen from a principally P-limited part of the system (the impoundment) into a principally N-limited part of the system (the estuary).

Working with data derived from southeastern lakes and reservoirs (predominantly reservoirs, extending from Mississippi to Maryland), Reckhow (1988) found the probability of blue-green algal dominance to increase with increasing TP, decreasing TN (and thus, by inference, decreasing TN/TP ratios), and increasing hydraulic retention

time. Blue-green algal dominance also tended to be associated with anoxic conditions in the hypolimnion.

### *Recommendations:*

When N and P concentrations are within the N limitation or the N-P co-limitation range (Table 4), nitrogen management can be expected to influence the algal community. Despite this fact, we see phosphorous management as a focal point of nutrient criteria controls because, in most situations, availability of P exerts primary control over algal populations. Although many Virginia impoundments exhibit N/P ratios within the co-limitation range (Table 4, Figures 6 and 7), tight relationships between P concentrations and algal indicators should not be expected. Pending the data analysis, we are not ready to recommend whether nutrient criteria would be established directly, as P concentration limits and/or indirectly as response variable (chlorophyll *a* and/or water clarity) limits.

We see no argument for establishing nutrient criteria for N that would be applied independently. At N/P ratios higher than the co-limitation range, it is unlikely that N reductions would have any effect if applied independently of reductions in P. At N/P ratios within the co-limitation range, it is possible that N reductions applied independently (not in association with P reductions) would shift the community composition towards greater representation by N-fixing bacteria, which would have a negative effect on the water bodies' suitability for the designated uses that nutrient criteria are intended to protect.

We do not see the establishment of N criteria in the form of an "ideal" N/P ratio for application in Virginia impoundments as logical or justified at this time. The major reason for this statement is the lack of a firm basis or justification for defining what such an ideal ratio would be in any individual impoundment or grouping of impoundments. No AAC members are algal specialists, but our review of the scientific literature indicates that scientific knowledge of how N and P levels act together to control community composition remains limited at best. Because Virginia's reservoirs and other impoundments were constructed and are not likely, therefore, to exhibit many of the water quality characteristics and interactions to be expected in natural systems, the use of natural systems as references for defining N/P ratio criteria would not be appropriate.

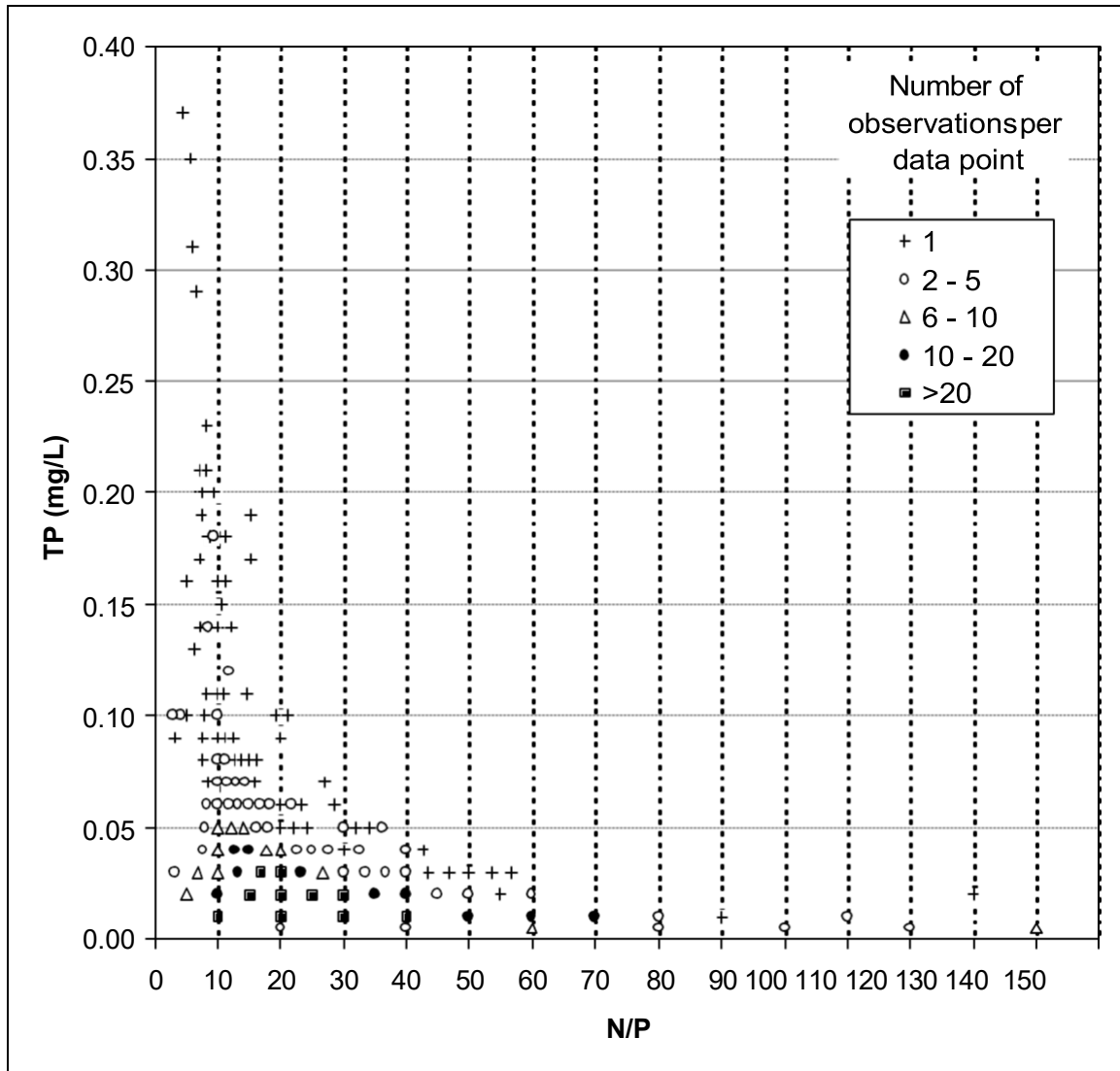
In establishing regulations and/or other guidance for defining how nutrient-criteria defined impairments would be remedied, DEQ should remain aware that concentrations of N, as well as P, will often influence algal populations. Whether or not this fact is of practical significance will be determined, in part, by the specific nutrient criteria that emerge from this process. We expect that these criteria will include either direct P concentration limits and/or response-variable limits that can be demonstrated to correspond roughly with P concentrations. Current data for Virginia's impoundments (Figures 6 and 7) indicate that, if the final criteria effectively limit P concentrations to 50  $\mu\text{g/L}$  (0.05 mg/L) or above, most or all potential impairments are occurring at relatively low N/P ratios. Therefore, such impairments would be best remedied by reductions of P inputs alone so as to reduce both algal levels and the potential for cyanobacterial blooms via the decrease of TP inputs and the resultant N/P ratio increase. If, on the

other hand, the final criteria effectively limit TP in the range of EPA guidance criteria (Part I, Table 1), then impairments in lakes where N and P are co-limiting would be more common. If such were to occur, there would be benefits to defining a structure for nutrient impaired waters that would provide incentives for reductions of N that can be achieved in association with P reductions without causing N/P ratios to decline, as opposed to placing sole emphasis on P reductions. Providing opportunities for trading N reductions versus P reductions would allow for more cost-effective restoration than would sole reliance on P reductions, but only in situations where water-quality conditions would benefit from N reductions. In applying such strategies, DEQ should remain aware of the potential for P release from sediments to delay water-column response to P loading reductions, although the effect of sediment-released P on summer algae can be expected to be minor in stratified reservoirs with low residence times (Marsden 1989).

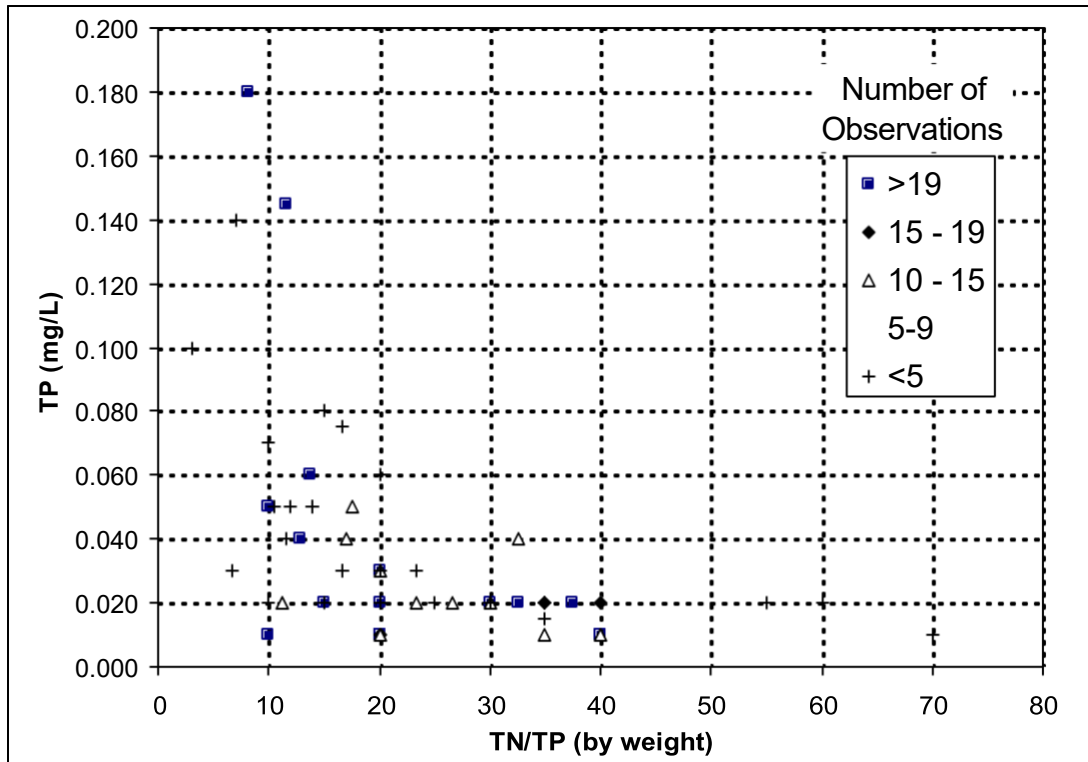
**Table 4.** Nutrient ratios (as Total N and Total P, by mass) and concentrations cited by various sources influencing algal mass and species composition, for reference in interpreting Figures 6 and 7.

Ratio or Level	Significance	Study
N/P > 22	At N/P ratios above 22, blue-green algal blooms seldom occur. Risk of bloom is increased below N/P = 22.	Smith et al. 1995
N/P = 10 to 15	Equilibrium N/P in systems where N fixers develop in response to N limitations	Hellstrom 1996
N/P < 10	Indicator of N limitation (ratio applied to inflow waters)	Flett et al. 1980 Hellstrom 1996
N/P = 7	“Redfield Ratio”	
N/P < 5	Indicator of N limitation	Matthews et al. 2002
5 < N/P < 20	Indicator of co-limitation by N and P	Matthews et al. 2002
4 < N/P < 23	Range of algal cellular N/P ratios	Suttle and Harrison, 1988.
Total inorganic N < 0.1 mg/L	Indicator of N limitation	Gophen et al. 1999
TN = 0.458, 0.358, 1.27 mg/L	EPA guidance criteria for TN in lakes, in Virginia’s western mountains, piedmont and western coastal plain, and eastern coastal plain respectively.	U.S. EPA 2000g
TP = < 30 µg/L	Risk of cyanobacterial dominance < 10%	Downing et al. 2001
TP = 30 – 70 µg/L	Risk of cyanobacterial dominance ~ 40%	Downing et al. 2001
TP ~100 µg/L	Risk of cyanobacterial dominance ~ 80%	Downing et al. 2001
TP = 50 µg/L	Current threshold used by DEQ for identifying freshwater lakes as “nutrient enriched waters”	
TP = 8, 20, and 17.5 µg/L	EPA guidance criteria for TP in lakes, in Virginia’s western mountains, piedmont and western coastal plain, and eastern coastal plain respectively.	U.S. EPA 2000g

**Figure 6.** DEQ monitoring observations for TP and TN/TP (by mass) ratios in Virginia lakes, 1990 – 2003, 0.3 m depth, April – October. For lakes with >20 samples, only the most recent 20 samples are represented. TN is calculated as a sum of measured components, with “lower-than-detection limit” observations calculated as ½ of the detection limit value. Note that TN/TP ratios (by mass) are approximate as TN concentrations can be calculated only to 0.1 mg/L precision, consistent with TKN analysis.



**Figure 7.** Median TP concentrations and TN/TP ratios for individual Virginia lakes, by number of observations per lake, calculated from the data used to plot Figure 6.



### Streams and Rivers

At this point in the AAC’s progress, we do not have a clear picture regarding whether or not criteria for streams and rivers should include TN, TP, or both. As with lakes, there is scientific evidence for co-limitation in rivers and streams by both N and P. This evidence is discussed, in a general sense, by Dodds and Welch (2000), p. 188: “Control of P alone may cause P to limit or lower algal biomass. ... However, if pulses of P occur, they can be taken up in excess of requirements and stored inside algal cells in a process called luxury consumption. This stored P can allow algae to grow even if P concentrations are low in the water column. If controlling such P pulses is impossible (e.g., pulses associated with high runoff events in the spring), control of N could become necessary.”

Dodds et al. (1997, 2002) conducted longitudinal studies of factors influencing benthic algal biomass in streams using databases derived from their own research, and from scientific literature describing studies conducted in temperate climates worldwide including North America, Europe, and New Zealand. In these studies, both TN and TP were found to exhibit significant correlations with benthic chlorophyll *a*. In two of the three analyses (the 2002 study used two separate databases), TN exhibited a stronger relationship with algal biomass indicators than TP. Dodds et al. (2002) state their expectation that “a pattern of co-limitation will become evident” in streams, as has been

observed in lakes. They note that the existence of widespread co-limitation in lakes is explained by the presence of non-equilibrium conditions and mixed species assemblages, both of which are also widespread in streams. In lakes, the potential for N loading reductions to stimulate nuisance algae populations (cyanobacteria) combined with the distribution of N:P ratios in lakes monitored by DEQ support the AAC's logic in recommending against establishing nutrient criteria that include only N controls. We have not yet reached any conclusions regarding whether or not similar logic should be applied in developing nutrient criteria for rivers and streams.

#### **8. Should narrative translators be expressed as percentages or other statistical factors or ratios?**

A narrative translator is a term that describes additional calculations used to better relate a measurable numerical criteria to a narrative standard or to a desired numerical criteria. For example, if a trace metal in a wastewater effluent may be measured in several ways, then a numerical criterion may be associated with the "total recoverable metal." From an ecological/environmental perspective what might be of interest is the dissolved metal. To formulate criteria in terms of dissolved metal, a translator is required. In this case, the translator is a fraction and is equal to the dissolved metal concentration divided by the total recoverable fraction.

In the case of nutrient criteria, the question of what type of translator depends on the criteria. Certain chemical standards are likely to use a proportional translator. For biological standards it seems that the criteria might involve statistical translators (for example, a trophic index). The statistical translator would relate concentrations of nitrogen and phosphorus to direct indicators of eutrophication.

### **Regionalization**

#### **9. Should Virginia consider adoption of ecoregion and water body type specific criteria developed by neighboring states with shared waters?**

It was observed that the best of all possible situations on shared waters would be the development of similar (preferably identical) criteria between neighboring states, and Virginia certainly should consider such an approach. There would be a need, however, for a detailed review of the criteria development process in the neighboring state(s) to ensure that common criteria would be in Virginia's interest, and for some review or analysis to determine that application of that shared criterion would be appropriate for Virginia.

It would appear that an ideal opportunity for application of shared criteria is presented by Lake Drummond and Mountain Lake, if neighboring states contain natural lakes with similar characteristics and have conducted an appropriate criteria development process.

Interstate partnerships have been shown to work in the development of appropriate and attainable criteria. A recent case between Oklahoma and Arkansas over a P criterion illustrates the value of interstate coordination.

## Classification

### **10. Should criteria development be broken out into water types: streams, lakes, and estuaries?**

The committee recommends that the criteria development process should be separated by water-body type: streams, lakes, and estuaries. In addition, natural and man-made lakes should be treated separately.

### **11. Should water body and depth-specific dissolved oxygen criteria be considered? In waters that experience dissolved oxygen deficiency, should dissolved oxygen be added as a response variable? Ex: State might demonstrate via a use attainability study that in a deepwater reservoir some phosphorus enrichment may be consistent with a particular game fishery designated use. A model might indicate that TP & DO adequately protect the deep reservoir or lake's designated uses and chlorophyll a is not required as an independent criterion.**

The question posed to the Academic Advisory Committee is to determine the need for and advisability of using dissolved oxygen (DO) as a response variable associated with the setting of nutrient criteria. In addition, if DO criteria are deemed advisable, how might the criteria be applied to whole-lake and/or hypolimnetic DO? The question is a logical extension of the DEQ Technical Advisory Committee report of 1987, which recognized the relationship between nutrient inputs and DO concentrations in lakes and reservoirs and suggested the possible need for establishing DO criteria in conjunction with nutrient criteria.

Unlike chlorophyll *a* and Secchi depth, which are primary response variables to nutrient inputs, DO is a secondary response variable. Increased nutrients lead directly to greater algal growth and decreased water clarity. The death and subsequent decomposition of algal cells then leads to the lowering of DO concentrations, with the primary concern being for the hypolimnion during stratified periods. Maintaining adequate DO in the water column is critical for aquatic life, and hence also has implications for the recreational use of water bodies, as well as having potential effects on the taste and odor of drinking water. The availability of DO in the hypolimnion and especially at the sediment-water interface also affects chemical reactions involving P, Fe, and S. For example, anaerobic conditions cause increased release of chemically bound P, which in turn increases the availability of P throughout the water column upon lake turnover. A positive feedback therefore occurs involving increased P concentration, greater lake productivity, and decreased DO concentrations. It should also be noted that short-term disturbances of mid-summer stratification may occur in

long, narrow reservoirs during periods of high storm event flows. Such stratification disturbances may result in the transport of high concentrations of soluble nutrients into the epilimnetic zone during the summer growing season.

The use of DO as a response variable thus is supported by the effect of nutrients on DO, and it in turn potentially affects the designated uses of lakes and reservoirs (aquatic life, recreation, drinking water). There are, however, a number of problems and disadvantages associated with using DO as a response variable and setting DO criteria. While there certainly is a direct, though secondary link, between nutrient and DO concentrations, DO also is affected by a number of other factors that confound the establishment of highly correlated relationships between nutrients and DO. Among these confounding factors is lake morphometry, water temperature, organic loading to the lake, inorganic turbidity, and at times, natural aeration rates. These and other factors combine to cause natural variation in DO concentrations among water bodies and especially at different depths that can mask the response of DO to nutrient concentrations.

In the case of lake morphometry, the shape of the lake “basin” itself may have a direct impact on productivity. For example, lakes of similar total volume, but having different ratios of epilimnetic volume to hypolimnetic volume (E:H) may be expected, for a given nutrient input, to have a different trophic response. It may be helpful to think of the epilimnetic volume as the principal zone of primary production in the water body. If this volume is large relative to the hypolimnion, then the organic detritus being deposited from the (larger) epilimnion to the (smaller) lake bottom may be expected to result in a higher rate of deoxygenation than that experienced in a low E:H reservoir.

Although maintaining adequate DO concentrations in the mixed layers of lakes and reservoirs is important, including in the epilimnion during stratified periods and throughout the entire water column during mixing periods, the primary focus on DO is on hypolimnetic oxygen deficits. There is a long limnological history of relating epilimnetic total P concentrations to primary production and then to DO in the hypolimnion. In order to use hypolimnetic DO as a response variable relevant to nutrient criteria, a number of points must be addressed.

- The designated use(s) of the water bodies must be determined, to include incorporating potential trade-offs among possible uses.
- There must be a setting of a threshold hypolimnetic DO concentration below which each designated use would be impaired. The designated uses of swimming and drinking water require a less strict threshold than does support of aquatic life, and even within the aquatic life use there can be considerable variability depending on the target aquatic life (e.g., cold versus warm water fish).
- Probably the most difficult consideration is that there must be a determination of the spatial (usually volumetric) extent of the hypolimnion that must be above the threshold DO concentration in order to support the designated use. It may be acceptable for a portion of the hypolimnion to be below the threshold DO concentration if the remainder of the hypolimnion is of sufficient size to support that use. An obvious example is related to the need of some fishes for well-oxygenated and cold (and hence deep) waters during the summer stratified

period. A portion of the hypolimnion may have inadequate oxygen, but the need for these fish may be met if a portion of the hypolimnion meets their temperature and DO thresholds.

- Many lakes undergo a level of hypolimnetic DO depletion associated with natural eutrophication as opposed to cultural eutrophication. An understanding of the background DO depletion or “reference” condition for each lake thus is necessary for setting DO criteria associated with nutrient criteria.
- Other aspects that should be considered in setting hypolimnetic DO criteria include the timing, frequency, and length of inadequate DO relative to the impact on designated uses.

Thus, if implemented, criteria for DO need to be targeted to specific designated uses, have thresholds that meet those uses at appropriate times of the year and for appropriate lengths of time, and be indexed to the extent to which the hypolimnion is affected (i.e., an areal or volumetric hypolimnetic DO deficit) relative to the needs of the designated use. If fish are the target designated use, for example, then a DO criterion may need to incorporate a sufficiently sized hypolimnion that meets not only the DO but also possibly the temperature requirements of the fish.

It also is important to remember that DO is a secondary response variable associated with nutrient concentrations. Before DO criteria can be established for designated uses, there needs to be developed a strong relationship between nutrient concentrations, primary production, and DO concentrations, as well as incorporating information on the extent of the hypolimnion affected. It is important to note, however, that setting hypolimnetic DO criteria and compensating for the size of the hypolimnion does not necessarily provide a value highly correlated with the productivity or nutrient status of the lake because the rate of DO consumption also is dependent on other factors, in particular water temperature and also organic loading, turbidity, and reaeration rates based on lake depth, strength of the thermal stratification, and water inflow to the hypolimnion.

Hypolimnetic DO concentrations are related to the nutrient status of lakes and reservoirs and are important for meeting various designated uses. It is difficult, however, to directly correlate nutrient status to DO concentrations within a viable framework to protect designated uses because of the variety of additional factors that must be considered. Using hypolimnetic DO concentrations as a response variable for nutrient concentrations, and setting accurate, usable, and defensible DO criteria, will require considerable effort. Effort in setting nutrient criteria should focus on other response variables such as chlorophyll and Secchi depth.

**12. Should Virginia utilize “use attainability” studies to refine uses, especially for lakes with multiple uses, such as promoting a game fishery while maintaining water clarity that promotes recreational swimming or should Virginia focus on determining appropriate, possibly more stringent criteria for a lake or reservoir that has a public water supply designated use?**

The committee sees use attainability analysis (UAA) as a valuable tool in criteria implementation. Application of UAAs is especially appropriate to constructed impoundments, such as reservoirs, because these water bodies were constructed with the intent of specific purpose(s) or use(s). Developed criteria should be protective of those intended purposes.

One reason why we expect UAAs to be an important aspect of criteria implementation is because the criteria will embody relationships between resource characteristics and the capability to serve the designated use and therefore, are characterized by some level of uncertainty. This result is likely because the ecological processes governing these relationships are complex. In this context, UAAs can be used to refine the criteria to protect the uses of the state’s water resources and are especially helpful to protect the uses of reservoirs and other man-made impoundments.

It should be noted, however, that there might be situations in multiple-use impoundments where protection of one designated use may create a conflict with another. For example, many reservoirs serve both recreational fishery and public water supply purposes. Whereas the public water supply use would be best served by more restrictive criteria intended to place tight limits on algal populations, recreational fisheries may be better served by more moderate algal levels. The ultimate criteria should seek to balance such tradeoffs.

**13. Should user perception surveys at lakes or a literature survey of user perception of lakes be used in determining appropriate criteria in lakes and reservoirs?**

User perception surveys could be of value in criteria development for lakes or reservoirs where existing or potential recreational usage would be significant. It should be noted, however, that to conduct such surveys is not a trivial task. Worse, if such surveys are not done in a scientifically defensible manner (question design, pretest, sampling protocol, appropriate statistical procedures, etc.), the results may be, at best, meaningless, and at worst, confounding. Therefore, user perception surveys should be applied to the criteria development process only if resources can be made available to conduct the surveys in a scientifically defensible manner that includes pre-testing the survey and an analysis to define the relationship of survey respondents to the population that those respondents are intended to represent.

**14. What types of physical classification schemes should Virginia use for lakes (such as size) and streams (such as stream order)? Should Virginia set regulatory size thresholds for lakes and reservoirs that would eliminate from the population small lakes - such as agricultural ponds – and lakes and reservoirs without public access?**

Lakes and Reservoirs

The committee is not clear on the authority that DEQ has to use lake or reservoir size and/or public access status to exclude waters from regulation. It would be helpful for this point to be addressed by DEQ staff.

Assuming the statutory authority to discriminate by size, however, the respondents all seemed to think that exclusion of certain small water bodies from criteria would be a reasonable approach for water bodies such as

- Stormwater management ponds (site level)
- Agricultural ponds
- Small impoundments and watersheds under the control of a single owner (no public access)

Even small facilities, however, might be candidates for regulation if they were shown to have detrimental effects on downstream water quality. However, it is reasonable to assume that large regional stormwater management ponds would, at some point, be of sufficient importance to a community that they become indistinguishable (at least to the public) from other impoundments. It would, however, be problematic to establish criteria for such facilities that would preclude the appearance of nuisance conditions.

For larger lakes, physical classification schemes based on factors known to influence algal response to nutrient levels – such as retention time or flushing rate, and average depth – and original purpose (for all but the two natural lakes) are more important than size and public access.

The committee notes that U.S. EPA nutrient criteria documentation (2000f, p 3-1) defines lakes as being water bodies that are greater than 10 acres in size, and with mean retention times of 14 days or greater.

Rivers and Streams

Committee members suggest that stream size is an important discriminator, in particular because this would strongly affect the structure and function of biological communities. Given the geologic and physiographic diversity of the Commonwealth, however, watershed area should be considered as a more appropriate proxy for stream size than stream order. Ecoregion locations should also be considered as classifiers.

The current condition of the aquatic community in a stream should also be considered as a classifier, especially relative to the presence/absence of rare species. For example consider the contrast between the Shenandoah River and Clinch River, both of which occur within the same ecoregion. Unlike the Shenandoah system, the

Clinch hosts a wide array of endemic species, including fish and mussel species listed as “threatened and endangered” and “at risk.” Additionally, the Clinch River is considered as a priority biodiversity conservation resource at the national level. The fundamental differences in the nature of the biotic communities within these systems may be seen as a basis for treating them differently within the process of nutrient criteria development, even though they occur within the same ecoregion. For example, if a “reference” approach (“relatively undisturbed,” “least impacted,” dominant land use specific, or otherwise) were adopted for establishing nutrient criteria, there would be some logic to identifying a separate set of references for a basin such as the Clinch that contains a high diversity of rare and endangered endemic species.

It should be noted that several studies of in-stream nutrient-periphyton relationships failed to find significant ecoregion differences. Based on their longitudinal study of two separate databases that included observations from throughout the U.S., Dodds et al. (2002) concluded that any ecoregion effect, if present, was “weak.” Pan et al. (1999) did discriminate geographic differences among nutrient-periphytic chlorophyll relationships, but those differences did not correspond with the U.S. EPA (2001) ecoregions. Our review of scientific literature indicates that periphytic biomass responses to nutrient concentrations are likely to vary with factors that are representative of ecoregion differences (such as conductivity and stream gradient), but not necessarily with ecoregion boundaries directly.

#### **15. Should Virginia develop site-specific criteria for the two natural lakes in the state?**

The committee recommends that both Mountain Lake and Lake Drummond should have individual nutrient criteria developed. Both water bodies are very different from any of the constructed impoundments in the Commonwealth, and further, would not be expected to be similar to each other. Each is, in its own way, a unique resource.

There is justification for this view because scientific literature has demonstrated that the morphometric differences between natural lakes and reservoirs cause them to respond differently to nutrient inputs. Retention times are generally lower in reservoirs, and drainage-area / storage-volume ratios are generally higher for impoundments than in natural lakes (Kennedy 2002). In impoundments, algal-nutrient response characteristics can be expected to vary within the water body, as algal response to nutrient inputs is likely to be depressed by non-algal turbidity resulting from sediment influx near the tailwaters of most impoundments, and throughout impoundments with low retention times (Kennedy 2002). Such factors are not expected to be of significance in either of Virginia’s two natural lakes.

Although the general form of nutrient-algal relationships observed in natural lakes can also be expected to occur in impoundments, algal response models developed in natural lakes cannot be applied directly to impoundments. Working in four North Carolina reservoirs characterized by high levels of non-algal turbidity, Smith (1990) concluded that both summer mean algal biomass, and the relative cyanobacterial biomass, were lower than predicted by models developed in natural lakes. Additionally, a study of data from 1,300+ artificial and natural lakes from throughout the U.S. found

chlorophyll *a* relationships with both TP and Secchi depth to be far more variable in artificial lakes than in natural lakes (Canfield and Bachmann 1981). In analysis of data generated by the National Eutrophication Survey at over 500 natural lakes and impoundments, Soballe and Kimmel (1987) found that algal abundance per unit of phosphorous was greater in natural lakes than in impoundments.

Working with data from 44 southeastern lakes and reservoirs (predominantly reservoirs), Reckhow (1988) found TP to be only weakly correlated with median chlorophyll *a* levels ( $r = 0.328$ , as log forms), suggesting that relationships among nutrients and algal biomass tend to be weaker in constructed impoundments than in natural lakes. Reckhow's study found a high (negative) correlation between Secchi depth and TP ( $r = -0.728$ , as log forms), while the correlation between Secchi depth and chlorophyll *a* was minimal ( $r = 0.094$ ); he did not measure suspended solids or non-algal turbidity. Reckhow interpreted these data to indicate that some sediment-associated P was not bioavailable, while the TP values did serve as indicators of suspended sediment levels. A study of 94 Missouri reservoir impoundments found that SD clarity exhibited a stronger relationship with total suspended solids (TSS) than to chlorophyll *a* (Jones and Knowlton 1993). Two related Midwestern studies found that the presence of non-volatile suspended solids at high concentrations tended to depress chlorophyll *a* responses to TP, presumably by limiting light availability in the water column and by acting as a source of measurable but non-bioavailable P (Jones and Knowlton 1993, Knowlton and Jones 1993).

#### **16. Should Virginia consider percentage of wetted stream perimeter coverage of macrophytes as a criterion of nutrient enrichment?**

Assuming we are talking about emergent species rooted in hydrosols, macrophyte beds growing in the wetted perimeter of streams are extremely variable in coverage, and their presence depends on multiple factors. In most fast-flowing wadeable streams, macrophyte beds are restricted to islands, bars, or shallow areas along the edge in which the water table is close to the surface. Because the hydrosols in which the macrophytes are rooted are frequently anaerobic, nutrients are highly available to the plant roots, and the plants may exhibit luxurious growth regardless of the nutrient load in the water column (Wetzel 2001). Other factors such as current, water hardness, and light availability contribute to variability in macrophyte beds such that mapping vegetated areas over several years shows them be constantly shifting mosaics (Allan 1995).

### **Prioritization & Coverage**

#### **17. If criteria development is broken out into water types, should the efforts run sequentially or concurrently?**

In an ideal world, it would be preferable to develop criteria on different water-body types concurrently. Given the fixed deadlines that have been applied to this process by

EPA, a concurrent process would allow a more deliberative approach by extending the time between start and completion for at least some water types. This decision, however, must be taken with full knowledge of the impacts on the overall level of effort required, and with a clear understanding of the deficiencies and/or compromises that may result from attempting to support concurrent efforts in a resource-constrained environment. In addition to finances, human resources must also be considered. If the same personnel are to be involved with development of criteria for the various water-body types while maintaining a full load of other responsibilities, a concurrent process may not be the best approach.

The committee sees this largely as a resource issue. If DEQ sees policy or regulatory aspects to the question, these factors should also be considered by DEQ.

**18. If N and P criteria are developed, should they be limited to site-specific studies, such as TMDLs?**

The committee remains somewhat unclear on this question, as the logic appears to be circular. That is, if the purpose of nutrient criteria is to provide a decision support system for determinations of impairment, how could the criteria be applied only after a determination of impairment has been made?

The committee does recommend, however, that criteria implementation should be accompanied by an evaluation and refinement process that would be conducted in association with TMDL studies, as noted in Part I of this report.

### **Inventory of Existing Data**

**19. Are the existing data sufficient for DEQ staff to develop water body specific criteria?**

There are a number of reasons why the AAC believes that DEQ's existing monitoring data are inadequate (in the case of streams) or barely adequate (for constructed impoundments, and only if DEQ data are combined with data obtained from other agencies) for developing nutrient criteria on a statewide basis. These reasons are summarized below.

- The lack of linkages in ambient river and stream monitoring between physical-chemical water quality and biological monitoring data programs.
- The lack of a biological monitoring database for lakes and reservoirs.
- The lack of data on periphytic algal indicators.
- The fact that probabilistic monitoring of rivers and streams takes place in the spring and fall (not the critical summer season), and includes only a single observation of chemical parameters.

An additional inadequacy, relative to the needs for developing water-body specific criteria for lakes and reservoirs, is that most of these water bodies are monitored on 5-year cycles. DEQ also lacks basic data on the morphometric characteristics of many of the state's impoundments.

We make these comments on the inadequacy of DEQ's data resources for the challenge of nutrient criteria development reluctantly (i.e., fearing that they could be misinterpreted as criticisms). We have great respect for the agency's monitoring efforts and the professionalism of its monitoring program's supervisory staff. We realize that two fundamental realities have combined to create the current situation:

- The defined purposes for DEQ's monitoring program, both historic and current, do not include nutrient criteria development.
- DEQ's monitoring program is resource constrained, and faces major challenges in maintaining monitoring activities that are adequate to its statutory requirements under the Clean Water Act and Virginia General Assembly directives, given the resources provided by EPA and the state.

### **Planned Data Collection**

#### **20. Does DEQ staff need to conduct additional monitoring data or undertake literature surveys for default data?**

The response to Question 19 above applies to this question as well. Before providing additional response, two uncertainties must be resolved:

- Determination whether or not the proposed data analysis for lakes and reservoirs yields useful results.
- Development of an approach or strategy for criteria development for rivers and streams.

### **Data Needs**

#### **21. Should Virginia explore differentiation of chlorophyll a for phytoplankton vs. periphyton dominated streams and rivers?**

Critical interacting factors regulating algal production in all habitats include light, nutrients, and current (Wetzel 2001). It has been generally found that most of the algal production in moderate to fast-flowing wadeable streams can be attributed to periphyton rather than plankton (Allan 1995). Algal and cyanobacterial cells found in the water column in such streams are generally thought to have been derived from periphyton that has been sloughed from the substrate by a variety of forces. In slow-moving lowland streams, there is some evidence that a riverine plankton (potomoplankton) may develop and persist. Sources of these riverine algae are reservoirs, oxbow lakes, side channels,

streamside lakes, and other similar habitats. Larger, slow moving streams may be potentially acceptable habitats for plankton unless the stream water is turbid due to sediment load. In large, shallow, moderate-to-fast flowing streams (wadeable streams) where light penetrates to the bottom, periphyton production may be considerable. It seems pretty clear that measuring chlorophyll *a* only in wadeable streams in Virginia is not sufficient to assess the impact of nutrients on stream algae. DEQ should certainly explore monitoring periphyton in streams where appropriate.

## Appendix A. Lakes and Reservoirs Data Analysis Plan

To meet the objectives of the work plan for assisting DEQ with freshwater nutrient criteria, the AAC proposes to analyze DEQ monitoring and associated data. The primary purpose of the analysis will be to determine whether or not such data demonstrate linkages between water column nutrients (as represented by TN and TP concentrations, chlorophyll *a* as an indicator of algal populations, and Secchi depth as an indicator of water clarity) and the suitability of water bodies to support designated uses, as defined by the Clean Water Act, in Virginia. A secondary purpose will be to provide additional information that will be supportive of DEQ's efforts to develop nutrient criteria in accord with EPA mandates.

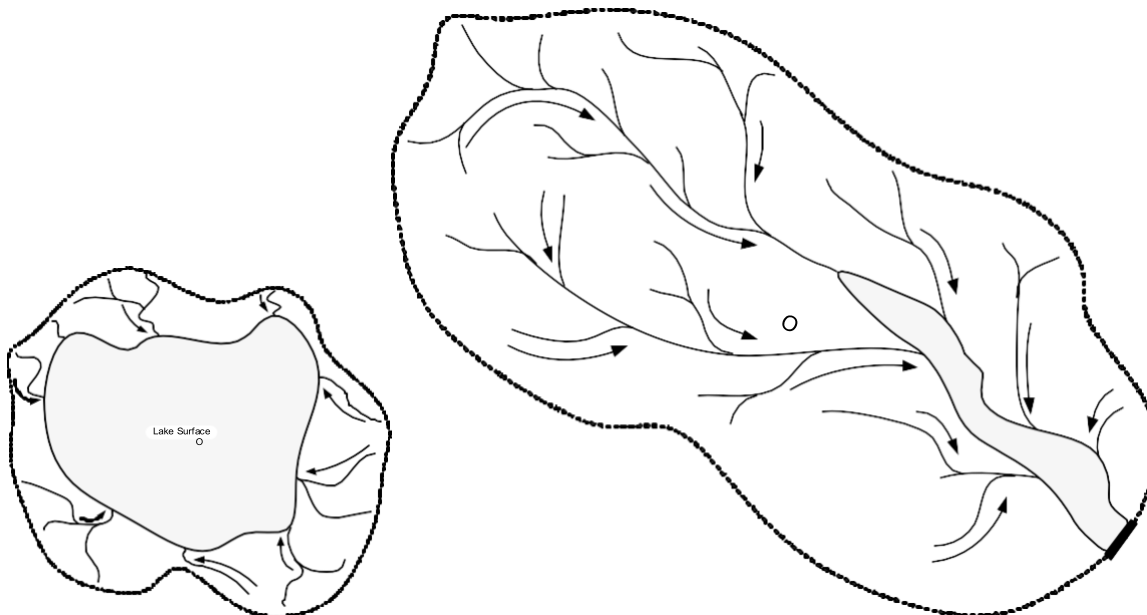
Virginia has two natural lakes and more than 100 constructed impoundments ("reservoirs") that have been monitored by DEQ. The AAC recommends that natural lakes be treated separately from reservoirs for the purpose of nutrient criteria development. Therefore, although the term "lake" is used throughout, the text that follows applies only to the state's constructed impoundments.

Kennedy (2002) notes that impoundments differ systematically from natural lakes in several ways that influence nutrient criteria development, including the fact that constructed impoundments generally have larger watersheds than natural lakes, and therefore exhibit greater influence by non-algal turbidity on phytoplankton growth. Also, the location of the water release mechanism on the impoundment structure will influence temperature regimes, creating a source of variability that is not present in natural lakes. For example, release of cooler water from a lower depth will result in warmer in-reservoir temperatures, while a surface release will cause the opposite effect. The location of individual monitoring points within the impoundment will determine the relative influence of these effects.

Physical, chemical, and biological differences in 309 natural lakes and 306 constructed reservoirs were compiled by Cooke and Carlson (1987), citing data from Walker (1981), and are summarized in Table A-1. As may be seen from the table, there are some striking differences, most of which may be related to the "human" purposes that reservoirs are generally constructed to serve. For example, reservoirs constructed for water supply and flood control are often sited to maximize drainage area, giving rise to the general characteristic, as noted above, of reservoirs having much larger watersheds than natural lakes. While achieving some engineered purpose, this trait also exposes them to higher mass fluxes of constituents (including nutrients) carried in tributary streamflows. As may be seen in the table, this was also reflected in the substantial differences in unit surface area loadings of both nitrogen and phosphorus. In fact, reservoirs were found, on average, to exhibit areal phosphorus loading rates over 3 times greater than those of natural lakes. In the samples compiled by Cooke and Carlson, the ratio of drainage area:pool area was almost five times larger for constructed impoundments. Another consequence of having relatively large storage volume to drainage area ratios is that natural lakes tend to have much longer hydraulic detention times than reservoirs. This gives rise to differences in the so-called "flushing rate" that is often used as a predictor of trophic response to nutrient loads. The drainage and pool area differences are illustrated schematically in Figure A-1.

**Table A-1.** Comparison of characteristics of natural lakes and reservoirs, after Cooke and Carlson (1987).

Variable	Natural Lakes (N=309)	Reservoirs (N=306)	Probability Means are Equal
Drainage Area (km <sup>2</sup> )	222.0	1358.0	<0.0001
Surface Area (km <sup>2</sup> )	5.6	8.6	<0.0001
Maximum Depth (m)	10.7	15.8	<0.0001
Mean Depth (m)	4.5	5.7	<0.0001
Hydraulic Residence Time (yr)	0.74	0.23	<0.0001
Drainage Area(km <sup>2</sup> )/Surface Area(km <sup>2</sup> )	33.0	156	<0.0001
Transparency (m)	1.4	1.2	<0.0005
Total Phosphorus (ug/L)	0.054	0.053	<0.02
Chlorophyll a (micrograms per liter)	14.0	10.0	<0.0001
P loading (gms P per m <sup>2</sup> per year)	0.87	2.9	<0.0001
N loading (gms N per m <sup>2</sup> per year)	18.0	45.0	<0.0001



**Figure A-1.** Schematic illustration of watershed and pool area relationships in natural lakes (left) and constructed impoundments (right).

## Available Data

### Water Quality

Virginia DEQ has provided the AAC with ambient monitoring data taken from the state's lakes since the late 1970s. Parameters include phosphorous concentrations (Total P [TP], ortho P), nitrogen concentrations (nitrate N, nitrite N, ammonium N, and total Kjeldahl N), chlorophyll *a*, and Secchi depth (SD), as well as context variables such as sampling location (DEQ monitoring station), sampling date and time, sampling depth, water temperature, and pH. Because some lakes are represented by multiple sampling points, DEQ has also identified the lake represented by each sampling point.

The AAC has also obtained from DEQ additional parameters for these water quality observations, including total suspended solids, volatile and/or non-volatile suspended solids, and electrical conductivity.

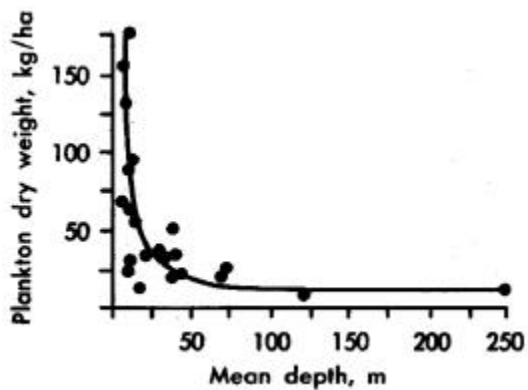
The AAC requests that DEQ identify those lakes that are treated routinely by copper sulfate, as such treatment will influence epilimnion nutrient-chlorophyll *a* relationships.

### Lake Characteristics

DEQ has provided data to the AAC defining the lakes' physical parameters (drainage area, surface area, volume, mean and maximum depths, retention time), primary use, and ecoregion location. The physical parameter data set is incomplete, but the AAC is working with DEQ in an effort to complete that data set for all lakes to be included in the data analysis:

- Virginia Department of Conservation and Recreation's (DCR) Division of Dam Safety maintains a database that includes selected physical parameters, including volume estimates. DCR has been requested to provide these data.
- Missing retention times will be estimated based on watershed area to volume estimates, adjusted for average annual rainfall.
- Where possible, estimates will also be made of the ratio of epilimnetic (E) to hypolimnetic (H) volume in lakes that thermally stratify.

The E:H ratio may be a good *á priori* indicator of trophic state, because it is reflective of the relative volume relationship between the zone where most algal production takes place (epilimnion), and the zone where decomposition processes are dominant (hypolimnion). If the ratio is high, a more productive trophic state may be anticipated for a given nutrient loading than if the ratio is low. The role that morphometric characteristics have in affecting trophic state is often underestimated. Figure A-2, reproduced from Cole (1994) and prepared using data from Rawson (1955), illustrates the relationship between planktonic production and mean depth in lakes. The figure supports a "rule of thumb" that, absent unusual anthropogenic nutrient sources, deep lakes are generally less productive than shallow ones, and that the boundary for oligotrophic systems is a mean depth of about 18 meters.



**Figure A-2.** Relationship between planktonic production and lake mean depth, after Cole (1994).

### Suitability for Aquatic Life

In general, impoundments in Virginia have been constructed for specific purposes such as water supply, recreation, and flood control. However, these impoundments must also meet other designated uses, including the support of aquatic life. The AAC will consider fish populations suitable for recreational fishing to be the primary biotic indicator of the capability of these water bodies to support aquatic life. Jensen et al. (undated manuscript) cite several studies that they interpret as suggesting that indicators of biotic integrity in natural waters are not appropriate for determining biological integrity in constructed reservoirs. Dr. John Ney is working with fishery biologists with the Virginia Department of Game and Inland Fisheries (VDGIF) to rate the status of each water body's fishery on a scale of 1 to 5, and to classify each fishery as one of three primary fishery types that are present in Virginia: warm water, cool water, and cold water.

The scale of fishery status evaluation is: How well does the water body support desirable species that achieve good growth and attain desirable size?

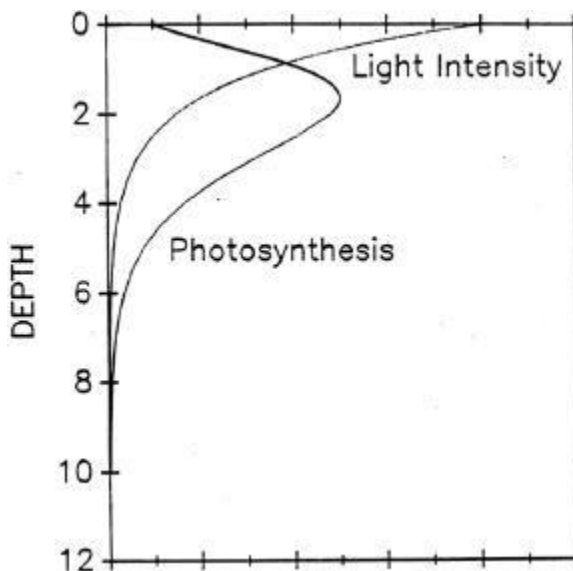
- 1 = poor: VDGIF biologists would recommend that anglers avoid such lakes.
- 2 = fair: VDGIF biologists would recommend that anglers fishing such lakes not expect much in the way of fishing success.
- 3 = average: the lake supports an adequate fishery.
- 4 = good: VDGIF would recommend such a lake for fishing.
- 5 = excellent: VDGIF would highly recommend such a lake for fishing.

Based on discussions with DEQ, the AAC plans to conduct these analyses with the goal of defining candidate nutrient criteria that are protective of fishery status at level 3.

## Data Analysis

Data analyses will be conducted in accord with basic guidelines provided by U.S. EPA (2000g, Chapter 7). All monitoring locations are aggregated by lake. For each variable, all observations from each lake will be reduced to a single data point that is considered to be representative of that lake over the period of interest using statistical methods appropriate to the variable's distribution. Because algal impairment is a warm-weather phenomenon, only observations representative of the warm-weather season will be considered. Because most algal growth takes place in the epilimnion and because the bulk of DEQ epilimnion monitoring has occurred at 0.3 m (~1.0 ft.) depths, only observations from 0.3 m depths will be used in the proposed analysis.

The AAC recognizes, however, that the selection of a single depth for retrieval of an epilimnetic chlorophyll sample may be unduly simplistic given the complex relationships between light intensity and peak photosynthetic activity. Figure A-3 is a schematic illustration of the relationship reported by Cooke and Carlson (1987). As may be seen, peak photosynthesis may take place at a depth where light intensity is substantially below its maximum value, thereby implying the potential for inhibition of photosynthesis by very high light intensity. The schematic, while not quantitative, may be used as an illustration. The peak photosynthetic activity is found at a depth of 2 meters, and is 5 times the magnitude found at the surface and about 2.5 times the value at 0.3 meters.



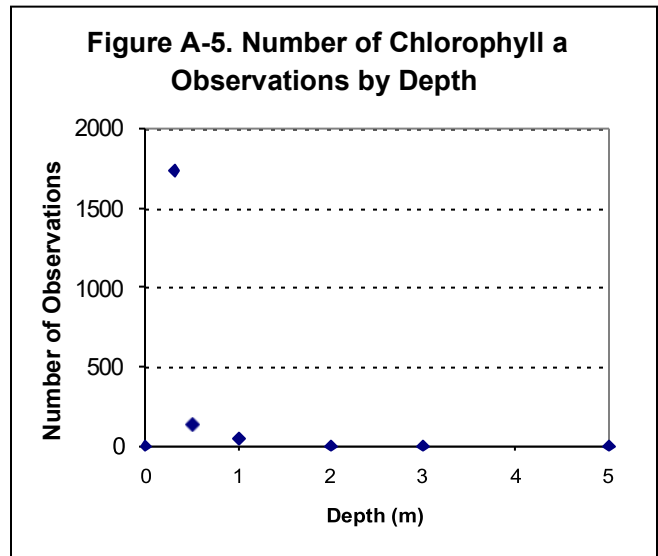
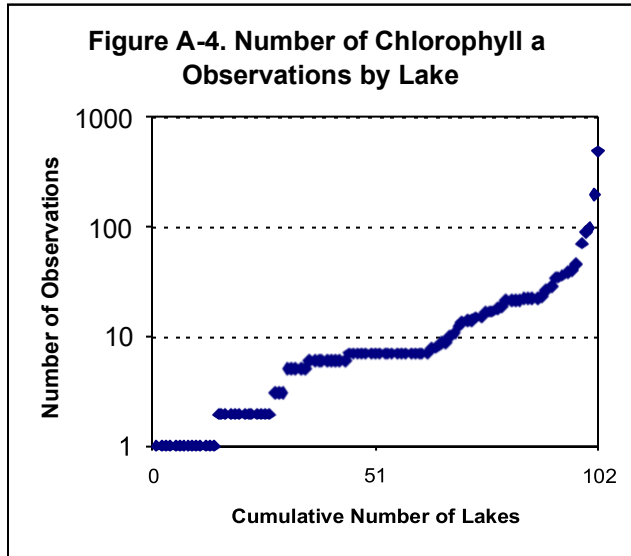
**Figure A-3.** Schematic depth relationship between light intensity and photosynthesis, after Cooke and Carlson (1987). Depth (vertical axis) is expressed as meters.

In accord with DEQ's discussion of variations in monitoring methods and data quality at the 6 January meeting with the AAC, only data from 1990 and later will be used in the analysis (Figures A-4 and A-5, Table A-2).

Total nitrogen (TN) values will be calculated by combining TN components using the

following rules.

- If nitrate-nitrite N is not measured directly, calculate as nitrate N plus nitrite N.
- If total N is not measured directly, calculate as nitrate-nitrite N + total Kjeldahl N.



**Table A-2.** Number of observations by variable at all locations, 0.3 m depth, by month (April - October).

Variable(s) Observed	Month						
	4	5	6	7	8	9	10
Chlorophyll a	134	194	264	327	330	196	221
SD	216	175	211	278	327	197	207
TN	303	305	367	429	461	256	339
TP	260	296	342	384	405	253	277
Chl-a, SD	95	108	162	238	248	174	139
Chl-a, TP	303	301	367	429	454	256	330
Chl-a, TN, TP	173	189	239	282	281	193	154
Chl-a, TP, SD	84	105	156	233	241	173	119
Chl-a, SD, TN, TP	84	105	156	233	241	173	119

## Questions

### 1. Do lakes' game fish populations vary with the lakes' nutrient status?

The primary goal of this analysis would be to determine the nutrient status that impairs the capabilities of reservoirs to support fish populations suitable for recreational fisheries. This analysis will be conducted by seeking relationships between fishery status (as defined by VDGIF biologists) and nutrient-related water quality variables. Based on published studies, fishery status can be expected to vary positively with algal biomass, and thus (by extension) with chlorophyll *a* and TP. In oligotrophic and mesotrophic systems, increasing TP can be expected to cause increasing algal biomass

and chlorophyll *a* and, thus, increasing fish populations. As nutrients and algal populations continue to increase through eutrophic and hypereutrophic conditions, fish populations can be expected to decline in response to the resulting conditions.

The AAC has obtained ratings of fishery status from VDGIF biologists for 60 lakes statewide, all of which are represented by five or more warm-weather epilimnion chlorophyll observations over the period extending from 1990 through 2003. Based on a preliminary analysis of the data, the AAC is concerned that the low representation of high-nutrient lakes may limit the statistical confidence of such a determination.

## *2. Do Virginia lakes demonstrate consistent relationships between water-column nutrient levels (TN and TP) and response variables (Secchi depth and chlorophyll *a*)?*

The form and variability of relationships between water-column nutrients (TP, TN) and response variables (chlorophyll *a*, SD) that are more directly associated with designated uses will have direct application to nutrient criteria development. Establishing criteria that include water column nutrient levels as well as response variables will be more direct and defensible if well-defined relationships between the two types of variables are found to be present.

Phosphorous is generally considered to be the limiting nutrient in freshwater lacustrine systems, and numerous studies have found strong relationships between total P and algal biomass (or its surrogate, chlorophyll *a*) in lakes (Schindler 1977, Schindler 1978, Canfield and Bachmann 1981, Smith and Shapiro 1981, Canfield 1983, McCauley et al. 1989, Correll 1998). The forms of algal responses to nutrient enrichment at relatively low levels are generally modeled as linear or log-linear functions. In studies including lakes with very high nutrient levels, several investigators have found TP-chlorophyll *a* relationships to be sigmoid (McCauley et al. 1989, Prairie et al. 1989), with reduced algal response to TP at higher concentrations. Such a response is consistent with what would be expected based on ecological theory because at higher P concentrations, other factors necessary for photosynthesis (micronutrients, sunlight...) are more likely to become limiting.

Many studies have found that TN concentrations, as well as TP, (or, in an alternative formulate, N:P ratio) also influence algal responses (Smith 1982, Canfield 1983, Smith 1983, McCauley et al. 1989, Prairie et al. 1989). Artificial enrichment experiments reviewed by Elser et al. (1990) demonstrate that enrichment by N and P in combination generally results in algal population increases that exceed those caused by equivalent enrichment by N or P alone.

Most nutrient studies were performed in natural lakes, and the application of their results to "artificial lakes" (created by impoundments) is not straightforward. One study of 1300+ artificial and natural lakes from throughout the U.S. found chlorophyll *a* relationships with both TP and Secchi depth to be far more variable in artificial lakes than in natural lakes (Canfield and Bachmann 1981).

Suspended solids concentrations tend to be higher in reservoirs than in natural lakes and can influence algal responses to water column nutrients. A study of 94 Missouri

reservoir impoundments found that SD clarity exhibited a stronger relationship with total suspended solids (TSS) than to chlorophyll *a* (Jones and Knowlton 1993). Two related Midwestern studies found that the presence of non-volatile suspended solids at high concentrations tended to depress chlorophyll *a* responses to TP, presumably by limiting light availability in the water column and by acting as a source of measurable but non-bioavailable P (Jones and Knowlton 1993, Knowlton and Jones 1993). Working with a data series consisting of 346 impoundments and 149 natural lakes from throughout the U.S., Soballe and Kimmel (1987) found non-algal turbidity to be greater, and algal abundance per unit phosphorous to be less in impoundments than in natural lakes. Kennedy (2002) also defines a factor [chlorophyll *a* \* SD], which he states can serve as an indicator of whether the light regime is more strongly influenced by non-algal turbidity or algal stimulation by nutrients. In reservoirs, the effect of non-algal suspended solids on nutrient-algal relationships in impoundments is generally considered to increase with decreasing retention times (Thornton et al. 1990, Chapter 6).

*3. Are influences by factors that may be used to classify lakes detectable in any of the above relationships?*

This analysis will be conducted because EPA guidance suggests that states should classify lakes during nutrient criteria development. The analysis will consider potential classifications that may influence nutrient criteria development. A primary potential influence would be the ecoregion, as the EPA recommends that nutrient criteria be established on an ecoregion basis.

Physiographic factors are also influential in reservoir nutrient-parameter relationships. Schindler's analysis of a data set of predominantly glacial lakes found water renewal time – a transform of retention time – to have a detectable influence on TP- chlorophyll *a* relationships (Schindler 1978). Soballe and Kimmel (1987) also found retention time to influence TP- chlorophyll *a* relationships in impoundments. In a study of Alabama reservoirs, Maceina et al. (1996) found retention time and mean depth, in addition to TP, to be positive and significant determinants of chlorophyll *a* concentrations. U.S. EPA (2000, Chapter 3) recommends that states consider three characteristics for use in categorizing reservoirs: location within the drainage basin, dam structure and operation, and hydraulic retention time.

Factors considered in this analysis will be determined, in part, by the morphometric variables contained in the data that has been requested from Virginia DCR. DCR is aware of the request and has responded positively with an intent to provide the data but, as of this writing, those data have yet to be made available to the AAC.

*4. Should Carlson's Trophic State Index (TSI) be considered as a scale for expressing Virginia's nutrient criteria?*

Trophic State Indices (TSIs) are indicators of the trophic status of lakes. A TSI is intended to represent a common scale, integrating measures such as water-column nutrients, algal concentrations, and clarity (U.S. EPA 2000, Chapter 2). Carlson (1977) developed a TSI that is widely used and is recommended by U.S. EPA (2000, Chapter

2) for consideration in nutrient criteria development. Carlson's TSI scale extends from 0 to 100. A TSI of close to zero represents an oligotrophic water body, while TSI's approaching 100 represent hypereutrophic status. Carlson presents formulas that can be used to calculate TSI from using any of three principal measures: TP, chlorophyll *a*, or Secchi depth transparency.

Carlson's TSI would provide the greatest benefit if TSI estimates calculated from TP, chlorophyll *a*, and SD were to demonstrate a high level of agreement. If such were to occur, it would be reasonable for DEQ to consider expressing nutrient criteria on a TSI basis. Divergences of TP, chlorophyll *a*, and SD TSI measures can also be useful in identifying water body conditions (U.S. EPA 2000, Table 3.2).

We will calculate TSI using measured TP, chlorophyll *a*, and SD values for selected Virginia reservoirs and quantify the degree of correspondence among these measures. We will explore the effect of expressing candidate criteria for protection of recreational and aquatic life uses as TSI values.

##### *5. What would be the implications of using the reference approach to establish nutrient criteria?*

The investigation of reference approach implications will be conducted because DEQ has requested that the AAC investigate the reference approach (Virginia DEQ 2003). However, discussions to date within the AAC indicate that most members do not see the "relatively undisturbed" reference approach as an appropriate basis for nutrient criteria to be applied to Virginia's constructed impoundments. This conclusion is based on the following observations:

- Virginia lakes are constructed impoundments. Therefore, the concept of an "undisturbed reference" is not reasonable.
- One purpose for constructing some of the smaller impoundments was for pollutant retention, and the larger reservoirs built in river channels perform this function. This function is especially important given the nutrient-sensitive status of Virginia's coastal receiving waters (e.g., Chesapeake Bay, Pamlico Sound). EPA documentation is specific in stating that states may consider impoundments' nutrient trapping functions in nutrient criteria development (U.S. EPA 2000, Chapter 7).
- The concept of an "undisturbed" or "minimally disturbed" reference has no relationship to designated use when applied to Virginia's constructed impoundments.

Nonetheless, we will consider three potential mechanisms for establishing reference conditions:

1. The ecoregion-specific guidance criteria communicated by EPA to the states in 2000.
2. Application of the "25<sup>th</sup> percentile" approach used by the EPA to promulgate its guidance criteria to Virginia lakes' data.

### 3. The 1987 TAC recommendations for defining “nutrient enriched” waters.

The effect of each potential reference criterion on potential nutrient impairments will be determined through hypothetical applications to the Virginia lakes’ data sets. The analysis will determine how effectively each reference criterion discriminates lakes with water-quality nutrient levels that support designated uses.

### **Ultimate Outcomes**

Based on the results of these analyses, the AAC will present candidate criteria for lakes that are protective of aquatic life.

The AAC will also fully define the context for these candidate criteria values by providing:

- An assessment of the strength of evidential support provided by data analyses and other scientific information for each of the candidate criteria.
- An assessment of how such candidate criteria can be best expressed (as water column nutrient concentrations, response variables, and/or TSI values).
- An assessment of the degree to which available data support reservoir classification using factors, such as ecoregion, and morphometric parameters, such as retention time.
- An analysis of how such candidate criteria relate to potential reference values.

## **Appendix B. Potential Threshold Values for Rivers and Streams from Various Sources**

Following are some parameter values that might be useful in setting thresholds for both the nutrient and response variables (See Tables B-1, B-2, and B-3). These values come from a variety of sources including research literature, analysis of data from select TMDLs regarding benthic impairments, and an analysis from some select reference and impaired streams in western Virginia.

**Table B-1.**  
**Potential Thresholds Based on Ranges of Previous Data**

**Potential Conservative Threshold Values Indicating a Clear Nutrient Impairment**

Data Description	No. of Sites	Samples per Site	Sub-Eco-region	Data Transform	NO3-N (mg/L)	TN (mg/L)	PO4-P (mg/L)	TP (mg/L)	Reference
Benthic TMDL Watersheds with upstream WWTP	4	23 - 72	67	a	2.20 - 3.24	2.52 - 5.86	0.046 - 1.390	0.069 - 1.636	1
Benthic TMDL Watersheds w/o upstream WWTP	3	12 - 72	67	a	1.27 - 2.27	1.48 - 2.58	0.021 - 0.024	0.029 - 0.092	1
Select sites with a benthic impairment	19	69	66,67,69	b	1.35 (0.40)	1.88 (0.94)		0.28 (0.10)	2
"threatened waters" threshold	--		All VA					0.20	3

**Potential Conservative Threshold Values Indicating a Clear Nutrient Non-Impairment**

Data Description	No. of Sites	Samples per Site	Sub-Eco-region	Data Transform	NO3-N (mg/L)	TN (mg/L)	PO4-P (mg/L)	TP (mg/L)	Reference
25th Perc. - Region 64	76 - 181	> 6,382	64	c	1.00	1.30		0.040	6
Estimate of max. natural GW conc.			DelMarVa		0.40				4
25th Perc. Of All Data, except Reg.64	5 - 650	> 1,014	VA, except 64	c	0.04 - 0.23	0.16 - 0.55		0.007 - 0.053	6
Select TMDL Reference Watersheds	4	7 - 44	67	a	0.27 - 1.20	0.51 - 1.51	0.015 - 0.029	0.012 - 0.071	1
75th Perc., Undeveloped Streams		63 - 82	Entire US	d	0.21	0.50	0.011	0.037	5
Select non-impaired biological reference streams	18	59	66,67,69	b	0.17 (0.14)	0.37 (0.38)		0.06 (0.07)	2

**Data Transformation Symbols:**

- a Average monthly ambient concentrations/site
- b Mean (median)
- c 25th percentile of all data/sub-ecoregion
- d 75th percentile of all undeveloped basins

For **Sub-ecoregion** designation, see Part 1 of this report, Figure 1.

**References**

- 1 - TMDLs developed for Benthic Impairments by the Biological Systems Engineering Department, Virginia Tech.
- 2 - Hill and Devlin, 2003.
- 3 - DEQ. 2002.
- 4 - Hamilton et al., 1993.
- 5 - Clark et al., 2000.
- 6 - EPA. 2000. Ambient water quality criteria recommendations. Information supporting the development of State and Tribal nutrient criteria.
  - Rivers and streams in Nutrient Ecoregion IX. EPA 822-B-00-019.
  - Rivers and streams in Nutrient Ecoregion XI. EPA 822-B-00-020.
  - Rivers and streams in Nutrient Ecoregion XIV. EPA 822-B-00-022.

**Table B-2.** Potential threshold values indicating nutrient impairment for chlorophyll *a* can be found in a number of compilation studies, including the following summary in EPA's *Nutrient Criteria Technical Guidance Manual* (U.S. EPA 2000g, Table 4). TN = total N, TP = total P, DIN = dissolved inorganic N, SRP = soluble reactive phosphorus.

**Table 4.** Nutrient ( $\mu\text{g/L}$ ) and algal biomass criteria limits recommended to prevent nuisance conditions and water quality degradation in streams based either on nutrient-chlorophyll *a* relationships or preventing risks to stream impairment as indicated.

PERIPHYTON Maximum in $\text{mg/m}^2$						
TN	TP	DIN	SRP	Chlorophyll <i>a</i>	Impairment Risk	Source
				100-200	nuisance growth	Welch et al. 1988, 1989
275-650	38-90			100-200	nuisance growth	Dodds et al. 1997
1500	75			200	eutrophy	Dodds et al. 1998
300	20			150	nuisance growth	Clark Fork River Tri-State Council, MT
	20				<i>Cladophora</i> nuisance growth	Chetelat et al. 1999
	10-20				<i>Cladophora</i> nuisance growth	Stevenson unpubl. data
		430	60		eutrophy	UK Environ. Agency 1988
		100 <sup>1</sup>	10 <sup>1</sup>	200	nuisance growth	Biggs 2000
		25	3	100	reduced invertebrate diversity	Nordin 1985
			15	100	nuisance growth	Quinn 1991
		1000	10 <sup>2</sup>	~100	eutrophy	Sosiak pers. comm.
PLANKTON Mean in $\mu\text{g/L}$						
TN	TP	DIN	SRP	Chlorophyll <i>a</i>	Impairment Risk	Source
300 <sup>2</sup>	42			8	eutrophy	Van Nieuwenhuysse and Jones 1996
	70			15	chlorophyll action level	OAR 2000
250 <sup>3</sup>	35			8	eutrophy	OECD 1992 (for lakes)

<sup>1</sup>30-day biomass accrual time

<sup>2</sup>Total Dissolved P

<sup>3</sup>Based on Redfield ratio of 7.2N:1P (Smith et al. 1997)

**Table B-3.** A collection of potential threshold values found in a review by Dodds and Welch (2000, Table 1).

TABLE 1. Various potential nutrient criteria set using different outcomes of concern related to instream nutrient concentrations. TN = total N, DIN = dissolved inorganic N.

Outcome	N (mg/L)	Total P (mg/L)	Comments
Toxicity, human	10 NO <sub>3</sub>		US national standard
Toxicity, aquatic life, acute	0.03-5 NH <sub>3</sub>		Fish and invertebrate data (Russo 1985)
Toxicity, aquatic life, chronic	0.005-1 NH <sub>3</sub>		Fish data (Russo 1985, Miltner and Rankin 1998)
Oxygen deficit, pH excursion	?	?	Probably greater than levels presented below
Mean benthic chlorophyll <50 mg/m <sup>2</sup>	0.47 TN	0.055	Large data set (Dodds et al. 1997)
Mean benthic chlorophyll <50 mg/m <sup>2</sup>	0.25 TN	0.021	Lohman et al. (1992)
Maximum benthic chlorophyll <200 mg/m <sup>2</sup>	3.0 TN	0.415	Calculated from Dodds et al. (1997)
Significant effect on biotic integrity index using invertebrates and fish	1.37 inorganic N	0.17	Headwater streams, Ohio (Miltner and Rankin 1998); effects less apparent in larger rivers
Systems with nutrient concentrations in upper ½	0.9 TN	0.04	Dodds et al. (1998)
Planktonic stream chlorophyll <8 µg/L	0.29 TN	0.042	Calculated from Van Nieuwenhuysse and Jones (1996); chlorophyll level from Organization for Economic Cooperation and Development (OECD, as cited in Rast et al. 1989); TN set by Redfield ratio (Harris 1986)
Lake mesotrophic/eutrophic boundary (planktonic chlorophyll <8 µg/L)	0.25 TN	0.035	OECD (as cited in Rast et al. 1989); TN set by Redfield ratio
Values set by State of Montana and co-operators	0.30 TN	0.020	Tri-State Implementation Council, Clark Fork Voluntary Nutrient Reduction Program
Levels leading to periphyton and macrophyte control	1.0 DIN	<0.020 (total dissolved)	Bow River, Alberta (A. Sosiak, Alberta Environmental Protection, personal communication)
Levels set to control summer phytoplankton		0.07	Tualatin River, Oregon (R. Burkhart, Oregon Department of Environmental Quality, personal communication).
Levels recommended to control maximum periphyton below 200 mg/m <sup>2</sup> for 50 d accrual	0.019 DIN	0.002 (soluble reactive)	(Biggs 2000)

## Appendix C: Influence of Nutrient Levels on Fish Populations in Lakes and Constructed Impoundments

Community energetics dictate that the biomass of fish at or near the top of the trophic pyramid should be highly dependent on the amount of primary production at the base (Lindemann 1942). Primary production in lakes is limited by nutrients, principally phosphorus. (*The Nutrient Criteria Technical Guidance Manual for Lakes and Reservoirs* [U.S. EPA 2000f] notes that nitrogen limitation is largely confined to subtropical and high altitude/latitude lakes).

Empiric relationships between fisheries productivity (as measured by fish harvest, production, or biomass) and both primary production and phosphorus concentration have been developed and published for regional and cosmopolitan sets of lakes. Correlations between primary production and fisheries productivity are highly positive, the former explaining ( $r^2$ ) 67%-84% of the latter (Table C-1). Correlations between total phosphorus (TP) concentration and fisheries productivity are equally strong (51% to 84%; Table C-2).

**Table C-1.** Predictive relationships between measures of plant and fish productivity in lakes and reservoirs, as determined from single-variable regression models.

Independent Variable	Dependent Variable	Data Set (n)	% of Variation Explained ( $r^2$ )	Source
Gross photosynthesis	Total fish yield	Indian lakes (15)	82	Melack (1976)
Phytoplankton standing stock	Total fish yield	Natural lakes, northern hemisphere (19)	84	Oglesby (1977)
Gross photosynthesis	Total fish yield	Chinese lakes and ponds (18)	76	Liang et al. (1981)
Chlorophyll <i>a</i>	Sport fish yield	Midwestern U.S. lakes and reservoirs (25)	83	Jones and Hoyer (1982)
Primary production	Total fish production	Cosmopolitan lakes (19)	67	Downing et al. (1990)

**Table C-2.** Relationship between total phosphorus concentration ( $\mu\text{g/L}$ ) as the independent variable and various measures of fish production in lakes and reservoirs.

Dependent Variable	Data Set (n)	% of Variation Explained ( $r^2$ )	Source
Total fish yield	North American lakes (21)	84	Hanson and Leggett (1982)
Sport fish yield	Midwestern U.S. lakes and reservoirs (21)	52	Jones and Hoyer (1982)
Total standing stock	Southern Appalachian reservoirs (21)	84	Ney et al. (1990)
Piscivore standing stock	Southern Appalachian reservoirs (11)	51	Ney et al. (1990)
Total fish production	Cosmopolitan lakes (14)	67	Downing et al. (1990)

Some of the above data sets were limited to natural lakes. Indeed, most of the analyses of trophic state (e.g., Carlson's TSI) are based on the relationships of phosphorus, chlorophyll *a*, and water transparency (Secchi disk depth) in northern natural lakes (U.S. EPA 2000). These relationships are less robust in reservoirs, which comprise 99% of Virginia's lentic waters. Chlorophyll *a* concentrations tend to be lower in reservoirs than in natural lakes (Soballe et al. 1992) because higher inorganic turbidity and flushing rates may limit the ability of phosphorus to stimulate phytoplankton production. In regression analysis of 80 southeastern U.S. reservoirs, Reckhow (1988) reported a weak relationship between chlorophyll *a* and phosphorus ( $r^2 = 0.10$ ), and virtually no correlation between chlorophyll *a* and transparency ( $r^2 < 0.01$ ). In these impoundments, inorganic turbidity largely determined water transparency, but the suspended sediment contained phosphorus; correlation of transparency with phosphorus was stronger ( $r^2 = 0.50$ ), although most of the phosphorus was not biologically available. Canfield and Bachman (1981) examined the National Eutrophic Survey (NES) data set and compared nutrient and response parameters between natural lakes and reservoirs. They found that, compared to natural lakes, reservoirs usually have substantially lower chlorophyll *a* concentrations for a given phosphorus concentration. Interpretation of their scatter diagram indicates that to produce  $10.0 \text{ mg/m}^3$  of chlorophyll *a* (indicative of marginally eutrophic conditions) in the average natural lake would require  $30 \mu\text{g/L}$  total phosphorus, whereas the average reservoir would require  $40 \mu\text{g/L}$  TP.

High flushing rates (low retention times) also limit development of phytoplankton

biomass. In fact, the *Technical Guidance Manual* (U.S. EPA 2000) recommends that

reservoirs with retention times less than 14 days be exempted from nutrient regulation because algal biomass buildup is minimal.

Because inorganic turbidity and flushing can limit nutrient impacts on reservoir productivity, it might be expected that the empiric relationship between phosphorus concentration and fisheries would be relatively weak. This does not appear to be the case in the southeastern U.S. Ney et al. (1990) examined the relationship between fish standing stock and a variety of potential predictors in a set of 21 southeastern Appalachian-region reservoirs for which fishery and water chemistry information was available for the same time frame (within 2 years). These reservoirs varied greatly in surface area (1,700 - 132,000 ha), retention time (4 - 438 days), and total fish standing stock (77 - 2,321 kg/ha). Total phosphorus was easily the best predictor of fish standing stock ( $r^2 = 0.84$ ), followed by Secchi disk depth (negative slope,  $r^2 = 0.42$ ) and chlorophyll *a* ( $r^2 = 0.31$ ). Fish standing stock increased linearly over the range of total phosphorus (8-81  $\mu\text{g/L}$ ), suggesting that maximum fish biomass would occur at higher phosphorus concentrations (Ney 1996). Fish production will ultimately be limited by habitat loss, resulting in a parabolic relationship with nutrient concentrations.

Total fish standing stock or total fish production may not be indicative of the sportfishing potential of reservoirs because sport and food fishes usually account for less than half the total. For the southern Appalachian reservoir data set, Yurk and Ney (1989) found that piscivore (largely game fish) standing stock increased linearly over the range of total phosphorus concentrations ( $r^2 = 0.51$ ). Jones and Hoyer (1982) reported that annual sportfish (synonym here for "gamefish") harvest increases linearly with total phosphorus over the range 15-90  $\mu\text{g/L}$  in 25 midwestern U.S. lakes ( $r^2 = 0.52$ ). In a study of 21 north temperate natural lakes, Hanson and Leggett (1982) found that long-term sport and commercial annual harvests increased with total phosphorus concentration up to 500  $\mu\text{g/L}$  ( $r^2 = 0.84$ ).

Individual species of sportfish are likely to respond differently to particular levels of lake fertility. The *Technical Guidance Manual* uses the work of Oglesby et al. (1987) to predict that as phosphorus in natural lakes increases, fisheries will shift from coldwater (salmonid) fisheries ( $P < 0.24 \mu\text{g/L}$ ) to percid (coolwater) fisheries ( $P = 24-48 \mu\text{g/L}$ ), and then centrarchid (warmwater) fisheries ( $P = 48-193 \mu\text{g/L}$ ). Total fisheries yield (harvest) will rise exponentially with phosphorus concentration. This progressive species shift was confirmed in an analysis of Minnesota natural lakes (Schupp and Wilson 1993).

The fisheries of Virginia's public reservoirs consist of a relatively few managed for trout (coldwater fishes), perhaps a third managed for a combination of coolwater (e.g., striped bass, walleye) and warmwater (black basses, sunfish, catfish) species, and the majority managed exclusively for warmwater fishes. For these systems to sustain quality fisheries, nutrient (phosphorus) management is critical; excessive nutrients limit habitat, while low nutrient levels limit food supply.

EPA's recommended approach to defining nutrient criteria equates the 25<sup>th</sup> percentile values for nutrients, transparency, and chlorophyll *a* for the set of all ecoregion lakes as target criteria. The approach is simplistic and based on scanty information (U.S. EPA 2000). The recommended criterion TP concentration for lakes and reservoirs in EPA's ecoregion XI is 8.0  $\mu\text{g/L}$ , which will support limited trout

fisheries and little else. Among subregions, the recommended TP concentration is as low as 5.0 µg/L. For ecoregions IX and XIV, the recommended TP criteria are 20.0 µg/L and 17.5 µg/L, respectively, which will support low-quality black bass and sunfish fisheries. Should these recommendations be adopted, fish production and the resultant socioeconomic benefits of Virginia's reservoir fisheries can be expected to decline sharply. Case studies of the response of reservoir fisheries to nutrient reductions (oligotrophication) support this scenario (Ney 1996, Stockner et al. 2000). Gains in habitat (oxygenated hypolimnia, reduced macrophyte stands) will likely be outweighed by loss of biological productivity.

The Clean Water Act (PL92-500) identifies protection and propagation of fish, shellfish, and wildlife as well as recreation (boating, swimming) as principal designated uses for lakes. A potential conflict is thus presented between maximizing fisheries productivity (especially for warmwater lakes) and accommodating recreational users. "Pea soup" lakes can be great fish producers but will be shunned by most anglers. Conversely, biologically sterile ("distilled water") conditions may be preferred by some non-anglers. The issue calls for compromise by addressing two questions: (1) How low must nutrient concentrations be to avoid undesirable plant production? and (2) How high must nutrients be to promote good fishing?

The answer to the first question depends on user perceptions. Although limnologists generally agree that TP of 30 µg/L can cause marginally eutrophic conditions in natural lakes (e.g., Secchi depth < 2 m), only 40% of 894 U.S. lakes and reservoirs met eutrophic criteria at TP = 40 µg/L (Walker 1988). Heiskary and Walker (1988) surveyed users of 99 Minnesota lakes, finding that impairment of physical appearances and swimming occurred in only 25% of those lakes studied at 40 µg/L TP. The *Technical Guidance Manual* (U.S. EPA 2000) offers little information on swimmable conditions: several states have 0.6 m to 1.0 m water transparency limits (to prevent drownings), which translate to 45-50 µg/L TP in natural lakes. The manual suggests that tolerance limits for boating are higher, equivalent to 90-100 µg/L TP. Extrapolation from studies elsewhere is an unsatisfying substitute for a Virginia user perceptions study.

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