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Technical Support Document for U.S. EPA's Final Rule for Numeric Criteria for Nitrogen/Phosphorus Pollution in Florida's Inland Surface Fresh Waters

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Disclaimer

This document is intended to support U.S. EPA's (hereafter EPA or the Agency) numeric water quality criteria for nutrients promulgated on November 14, 2010 pursuant to section 303(c)(4) of the Clean Water Act (CWA) (40 CFR §131.43). The information provided herein does not substitute for the CWA or EPA's regulations; nor is this document a regulation itself. Thus, this document cannot and does not impose any legally binding requirements on EPA, states, authorized tribes, the regulated community, or any other party, and might not apply to a particular situation or circumstance.

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Scope and Purpose of this Document

This document contains scientific information EPA used to derive numeric criteria for Florida's Class I and III inland fresh waters (streams, lakes, and springs) promulgated on November 14, 2010 pursuant to section 303(c)(4) of the CWA (40 CFR §131.43, hereafter final rule). EPA developed these criteria to support the State of Florida's designated uses for inland fresh waters from the effects of nutrient pollution. EPA's final rule follows its proposed rule for numeric nutrient criteria for Florida's inland flowing waters published on January 14, 2010 ("Water Quality Standards for the State of Florida's Lakes and Flowing Waters; Proposed Rule", EPA-HQ-OW-2009-0596, FRL-9105-1; hereafter proposal or proposed rule). EPA subsequently published additional information and requested comment on August 10, 2010 ("Water Quality Standards for the State of Florida's Lakes and Flowing Waters; Supplemental Notice of Data Availability and Request for Comment," EPA-HQ-OW-2009-0596, FRL-9185-2; hereafter supplemental notice).

For water quality management purposes, the State of Florida has designated the uses of Class I and III inland waters pursuant to the CWA (section 303(c)) and its implementing regulations (40 CFR 131). Class I waters are designated for "potable water supplies" while Class III waters are designated for "recreation, propagation and maintenance of a healthy, well-balanced population of fish and wildlife" (Rule 62-302.400, F.A.C.). Specifically, EPA has derived these numeric criteria to translate the State of Florida's existing narrative water quality standard for nutrients, applicable to these waters, at Rule 62-302.530(47)(b), F.A.C.:

> "In no case shall nutrient concentrations of a body of water be altered so as to cause an imbalance in natural populations of aquatic flora or fauna."

This document consists of chemical, physical, and biological data, meta-data, methods, and analyses EPA used to derive the final numeric nutrient criteria to translate the State of Florida's existing narrative criteria for Class I and Class III inland surface fresh waters (streams, lakes, and springs).

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Foreword

Excess nitrogen and phosphorus loadings, or nutrient pollution, are a leading cause of water quality degradation in the Nation's waters. Nutrient pollution may result in harmful algal blooms, reduced spawning grounds and nursery habitats, fish kills, oxygen-starved hypoxic or "dead" zones, as well as public health concerns related to impaired drinking water sources and increased exposure to toxic microbes such as cyanobacteria. Nutrient pollution problems can exhibit themselves locally or further downstream leading to degraded water quality in lakes, reservoirs, and estuaries.

One way nutrient pollution can be managed effectively is through the development and adoption, of numeric nutrient criteria into state water quality standards, and subsequent implementation of those water quality standards. State water quality standards are the foundation for protecting the quality of the Nation's surface waters and are the cornerstone of the water quality-based control program mandated by the CWA. Water quality standards describe the desired condition of a waterbody and consist of the following three principal elements: (1) the "designated uses" of the state's waters (e.g., fishing, aquatic life, drinking water); (2) "criteria" specifying the amounts of various pollutants, in either numeric or narrative form, that may be present in those waters that will protect the designated uses; and (3) anti-degradation policies providing for protection of existing water uses and limitations on degradation of high quality waters (See 40 CFR 131, Subparts A, B, C).

As noted in the Scope and Purpose of this Document, the State of Florida has defined the designated uses of its lakes and flowing waters as follows: "Potable Water Supply" (Class I) and "Recreation, Propagation and Maintenance of a Healthy, Well-Balanced Population of Fish and Wildlife" (Class III) (Rule 62-302.400, F.A.C.). The State of Florida's existing water quality criteria for nutrients are expressed as a narrative at Rule 62-302.530(47) (a) and (b), F.A.C. 62-301.530(47)(b) provides:

> In no case shall nutrient concentrations of a body of water be altered so as to cause an imbalance in natural populations of aquatic flora or fauna.

One advantage of numeric nutrient criteria compared to narrative criteria, once adopted and implemented, is that states can use the numeric criteria in quantitative state water quality assessments and watershed protection management activities. Numeric nutrient criteria support National Pollution Discharge Elimination System (NPDES) permits and total maximum daily loads (TMDLs) for nutrients. They can also be used in state- and community-developed environmental baselines to measure progress and support partnerships based on activities such as nutrient trading, land stewardship, and urban stormwater runoff control strategies.

Aquatic life water quality criteria consist of three key components: magnitude, duration, and frequency. Criteria must contain sufficient constituents to be protective of the designated use and must be based on a sound scientific rationale (40 CFR 131.11(a)). When setting water quality standards, the water quality standards of downstream waters must be taken into consideration to ensure that those downstream standards are attained and maintained (40 CFR 131.10(b)). In previously published guidance, EPA generally recommends that states develop and adopt numeric nutrient criteria for two causal parameters, such as total nitrogen (TN) and total

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phosphorus (TP), and two response parameters, such as chlorophyll *a* (chl-*a*) and clarity, into their water quality standards to protect against the effects of nutrient pollution.

For its final rule for inland surface fresh waters for the State of Florida, EPA has finalized numeric nutrient criteria for streams (TN, TP), lakes (TN, TP, chl-*a*), and springs (nitrate+nitrite). EPA derived criteria for these parameters based on the best available scientific data and information at the time. EPA also considered the existing water quality standards in the state (e.g., clarity). Where EPA has not finalized criteria for certain parameters in its final rule due to insufficient scientific evidence to support a precise threshold for numeric nutrient criteria (e.g., chl-*a* for streams), EPA or the state may consider deriving criteria in the future for those parameters.

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Chapter 1: Derivation of EPA's Numeric Nutrient Criteria for Streams

1.1 Introduction

Streams are unique and valuable ecosystems. They provide a wide variety of important and valuable services such as serving as habitat for a variety of fresh water species, including valuable recreational taxa; supporting a variety of recreational activities; and conveying drinking water. Florida is replete with these valuable ecosystems and the development of numeric nutrient criteria will assist in their protection and ability to support a balanced natural population of aquatic flora and fauna.

Nutrients are a natural component of stream ecosystems. In natural concentrations, which can and do vary, essential nutrients help maintain stream ecosystem structure and function. However, in excess quantities, nutrients can destabilize stream ecosystems, leading to a variety of problems including nuisance plant and algal growth, loss of physical habitat, hypoxia and anoxia, species loss, and toxins that can present a risk to recreation and drinking water (Dodds et al. 2009; Howarth 2002; NRC 2000; Smith 2003).

The adverse effects of excess TN and TP on streams are well documented (Biggs 2000; Bothwell 1985; Bourassa and Cattaneo 1998; Bowling and Baker 1996; Cross et al. 2006; Dodds and Gudder 1992; Elwood et al. 1981; Francoeur 2001; Gaiser et al. 2005; Moss et al. 1989; Mulholland and Webster 2010; Notestein et al. 2003; Peterson et al. 1985; Rosemond et al. 1993). However, spatial and temporal variability in environmental conditions influence the biological responses to excess nutrients. As a result, adverse effects can be described but not all of these effects are observed in every stream at all times. For example, some streams are shaded, which reduces the near-field algal growth response to excess nutrients because light, which is essential for plant or algae growth, is reduced at the water surface. In these streams, excess nutrients primarily increase rates of microbial activity and heterotrophic respiration in the near field. Other streams have rapid currents, reducing the time algae have to take up excess nutrients. However, when the same stream widens and slows downstream, or the canopy that provided shade thins, then excess nutrients accelerate plant and algal biomass production. Under such dynamic conditions algae can accumulate or be removed rapidly, making it difficult to capture in monitoring, and making biomass accumulation in an individual stream difficult to characterize. Due to these and other factors, the effects of excess nutrients may be subtle or dramatic, may be easy or difficult to capture by measures of plant and algal response (such as chl-*a*), and may occur in some locations along a stream but not others.

EPA recognizes that nutrient limitation of either TN or TP, or co-limitation by TN and TP, is a phenomenon previously observed (Schindler 1974). Primary production in fresh water systems can be limited by available nitrogen or phosphorus, or can be co-limited by both nutrients simultaneously. Recent studies have shown that nitrogen limitation and co-limitation are more important in fresh water, and phosphorus-limitation is more important in saltwater, than previously recognized (e.g., Elser et al. 2007; Smith 2003. For any given stream, however, it is difficult to assess *a priori* which nutrient is the limiting factor. EPA's approach to deriving numeric nutrient criteria is not premised on the limiting nutrient, but rather a conservative view that both nutrients influence excess algal growth over and that influence may vary over space and time. EPA has expressed its recommended approaches in deriving numeric nutrient criteria

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for both TN and TP previously for streams (U.S. EPA 2000) and as a general policy matter (Grubbs 2001; Grumbles 2007). The scientific literature concluding that the source of many adverse water quality impacts can be traced to degradation of water quality upstream of the impacted aquatic system by nutrient pollution (e.g., Conley et al. 2009; Paerl, 2009; U.S. EPA 2007) further affirms EPA's approach water quality management, through numeric nutrient criteria, of both nitrogen and phosphorus.

This chapter describes the methodology and results of EPA's analysis for deriving numeric nutrient criteria for Florida streams to protect designated uses in response to the environmental threat of nutrient over-enrichment. The criteria development process is summarized in EPA's analytical plan shown in Figure 1-1. A conceptual model of how nutrients affect designated uses in streams is shown in Figure 1-2.

To account for natural variability in nutrient concentrations of Florida streams, EPA organized streams into naturally homogenous classes based on geographic variation across the Florida landscape (geology) as well as differences in large-scale hydrologic features (watersheds). The process of classifying streams to account for and reduce natural variability is an important first step in nutrient criteria derivation (U.S. EPA 2000).

EPA applied a reference condition approach to derive numeric nutrient criteria for Florida stream using two populations of data to represent nutrient concentrations in Florida streams. The reference condition approach relies on identifying a least-disturbed set of reference sites reflecting "relatively undisturbed" conditions reflecting natural integrity (Davies 1997). These reference sites provide the appropriate benchmark against which to determine the nutrient concentrations present when the designated uses of a waterbody are being met. Setting criteria based on conditions observed in these reference sites reflects both the stated goal of the CWA and EPA's National Nutrient Strategy that calls for states to take precautionary and preventative steps in managing nutrient pollution to maintain the chemical, physical and biological integrity of the Nation's waters before adverse biological and/or ecological effects are observed (Grubbs 2001, Grumbles 2007).

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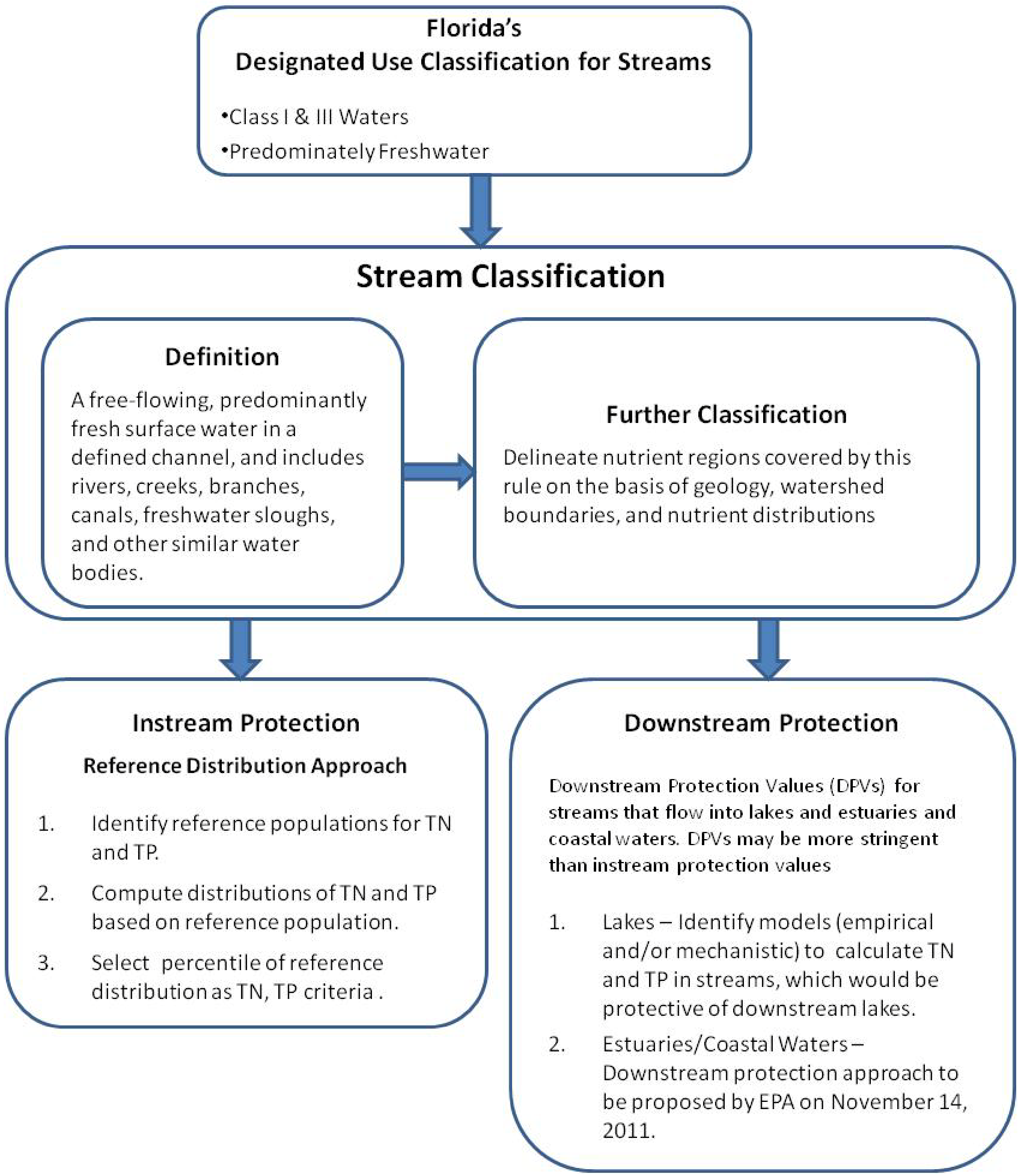

Figure 1-1. Analytical plan of EPA's numeric nutrient criteria development process for Florida's streams.

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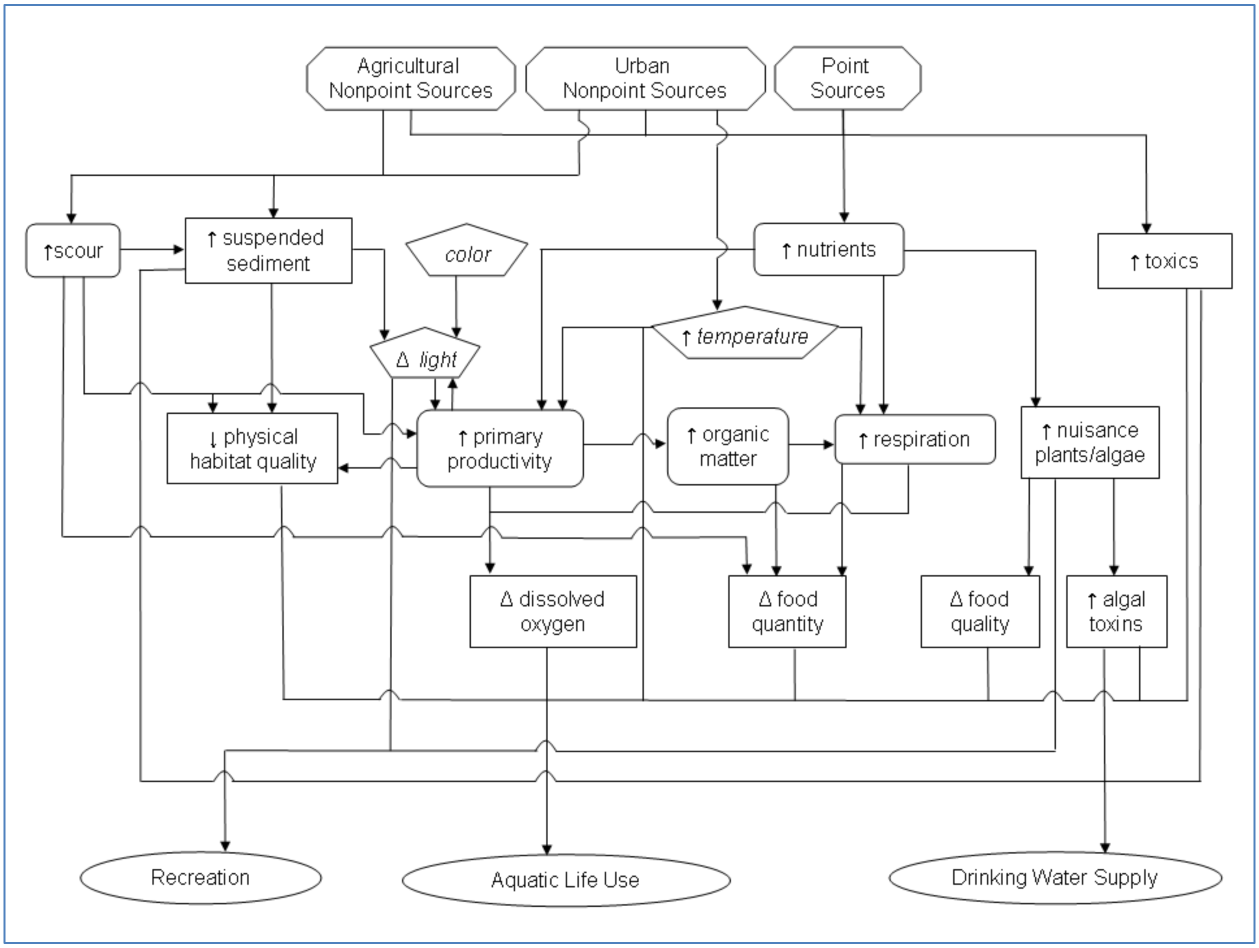

Figure 1-2. Conceptual model showing linkages between nutrient concentrations and designated uses in streams.

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EPA identified a population of least-disturbed reference sites (herein referred to as the Benchmark Population) using a method based on previous work by FDEP (FDEP 2009). That method was presented as an alternative for public comment in EPA's proposed rule (U.S. EPA 2010a) and was further described in EPA's supplemental notice (U.S. EPA 2010b). The method used to identify a least-disturbed population of sites, consisted of a set of screens based on land cover, aerial surveys, site visits, nitrate concentration, CWA section 303(d) impairments, and biological condition measures (Stream Condition Index or SCI). The sites were associated with waterbody identification numbers (WBIDs)[2], which are spatial units used by the FDEP for purposes of water quality monitoring and assessment. EPA also identified a reference population of biologically healthy sites (herein referred to as the Stream Condition Index [SCI] Population) using the approach detailed in U.S. EPA 2010a and considered further in U.S. EPA 2010b. The SCI Population was determined by identifying sites whose SCI score was greater than 40 and were not in WBIDs impaired for nutrients or dissolved oxygen. The SCI is a multi-metric macroinvertebrate index developed by FDEP to evaluate biological condition in streams; the threshold value of 40 and higher was determined to represent biologically healthy conditions in Florida streams.

1.2 Development of Nutrient Watershed Regions (NWRs)

Spatial frameworks are important for structuring research, assessment, monitoring, and management of environmental resources. Ecoregions are a common example of a spatial framework. Ecoregions may be defined by patterns of homogeneity in a combination of factors such as climate, physiography, geology, soils, and vegetation (Griffith et al. 1994). The development of regional nutrient criteria by definition balances site-specificity with regional applicability. In addition, there is understood natural variability among sites within regions with regards to nutrient chemistry due to natural biogeographic factors. Identifying regions to reduce natural variability improves the precision of criteria in setting realistic and protective expectations for a stream. Classification provides a framework on which to develop and base protective nutrient criteria, the goal of which is to account for and reduce the natural variability in nutrient chemistry due to geographic factors for any stream within a class. Proper classification identifies homogeneous populations of streams with similar nutrient regimes and biological communities. Classification helps to ensure that the thresholds selected from a benchmark or reference condition approach will be appropriately supportive of balanced natural populations of flora and fauna inhabiting the different classes. EPA nutrient criteria guidance recommends classification as an important step in developing protective nutrient criteria (e.g., U.S. EPA 2000).

Florida's geology includes sedimentary deposits of marine origin. Certain marine clays and limestone formations (e.g., the Hawthorn Group) that lie near the surface are high in phosphorus (Figure 1-3). Some of these phosphatic deposits are mined, making Florida one of the larger global producers of phosphate. Florida produces approximately 25% of phosphate used throughout the world. The natural phosphatic soils associated with the Hawthorn Group can also contribute to increased phosphorus concentrations in streams. For this reason, proper spatial classification is essential to capture regional differences in natural nutrient concentrations.

[2] A WBID is a spatial unit prescribed by the FDEP for purposes of water quality monitoring and assessment.

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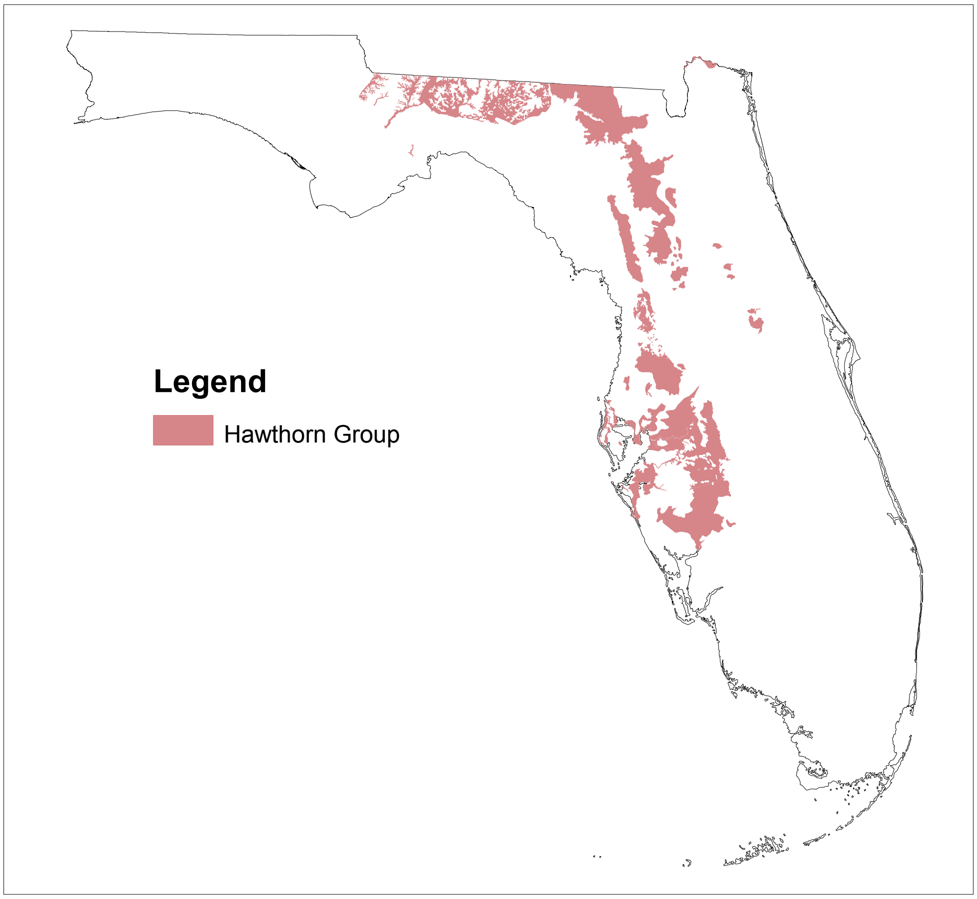

Source: FDEP (http://www.dep.state.fl.us/gis/, accessed in March 2010).

Figure 1-3. Map of Hawthorn Group in Florida, indicating major phosphatic regions.

1.2.1 Refinement of Regionalization for Nutrient Criteria Development

EPA considered previous bioregion work by FDEP (FDEP 2009) for purposes of classifying streams for nutrient criteria development. The four stream bioregions developed by FDEP were based on observed biological community similarity within a bioregion, assuming that biological responses to nutrients would be more similar within and not between bioregions. Evaluation of sites across bioregions indicated that the bioregions were not sufficiently homogenous with regard to nutrient concentration. FDEP also considered geological formations (e.g., the Hawthorn Group) and performed geostatistical analysis of ambient nutrient concentrations (e.g., kriging) for purposes of classifying streams for numeric nutrient criteria development. A sub-region of the Peninsula Bioregion—identified at EPA's proposal as "the Bone Valley" (subsequently renamed the West Central NWR for purposes of EPA's stream classification for numeric nutrient criteria development)—and which has high natural phosphate levels, was identified by FDEP during initial work to derive reference-based TMDLs for the Northern Lake Okeechobee tributaries. The naturally high phosphate areas occur in portions of Hillsborough, Polk, Hardee, Manatee, DeSoto, and Sarasota counties, where specific components of the Hawthorn Group, which includes the Peace River Formation and the Bone Valley Member

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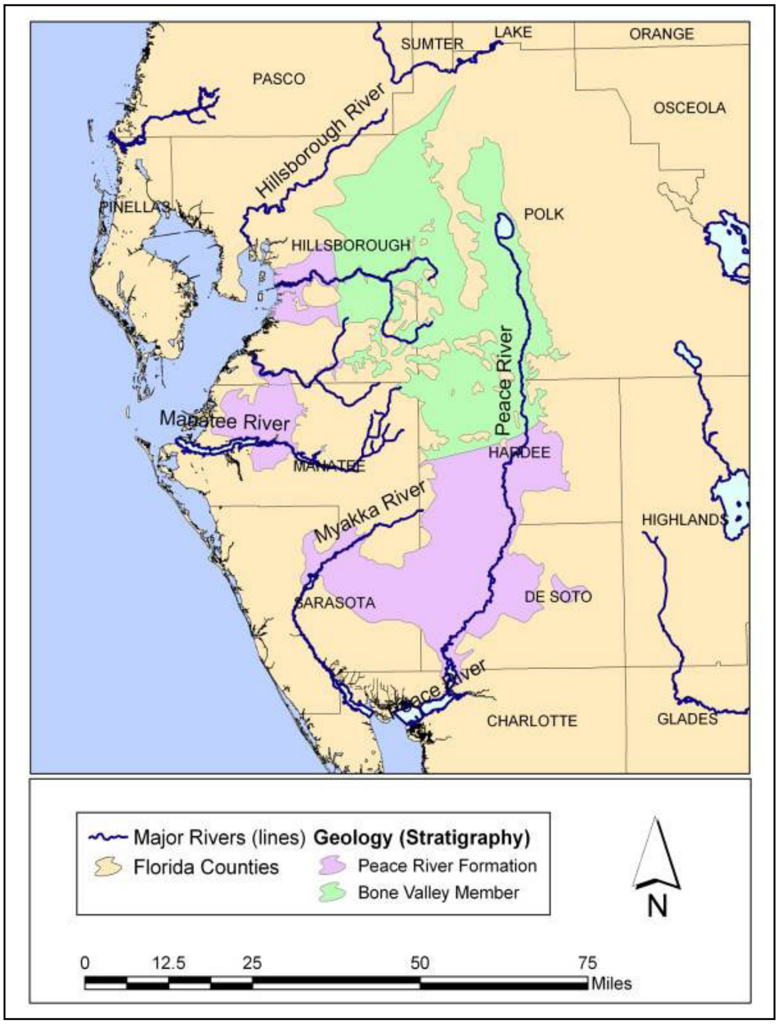
(originally the Bone Valley Formation of Matson and Clapp 1909, Figure 1-4), occur at the surface. The Bone Valley Member is a unique phosphate deposit that provided much of the phosphate production in the United States during the 20th Century. Mining of phosphate in the outcrop area began in 1888 (Cathcart 1985) and continues to the present.



Source: FDEP (2009).

Figure 1-4. Map of surficial geology showing the location of the Peace River Formation and Bone Valley Member.

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1.2.2 Development of the Nutrient Watershed Regions (NWRs)

EPA considered the previous work by FDEP on bioregions, nutrient regions, and sub-regions to develop a regional stream nutrient classification approach that addresses the natural variations in nutrient concentrations (including underlying geology) and reflects the understanding that upstream water quality affects downstream water quality. The resulting watershed-based classification enables EPA to address the effects of TN and TP within streams, as well as the effects of TN and TP from streams that discharge into downstream lakes or estuaries in the same watershed. EPA classified Florida's streams north of Lake Okeechobee, but including the Caloosahatchee drainages to the west of the Lake and St. Lucie and Loxahatchee drainages to the east, into NWRs. This was accomplished using WBID descriptions and verifying with drainage basin boundaries, both available from the FDEP GIS site (http://www.dep.state.fl.us/gis/datadir.htm, accessed March 2010). The resulting NWRs reflect inherent differences in the natural factors that influence nutrient concentrations in streams (e.g., geology, soil composition, hydrology).

For initial watershed regionalization analysis, EPA used nutrient water quality data collected across Florida and contained within Florida's Impaired Waters Rule (IWR) database. Data tables of nutrient water quality sampling stations and their associated WBIDs were joined to the National Oceanic and Atmospheric Administration (NOAA) Coastal Assessment Framework (CAF) delineation of estuarine drainage areas (EDAs), fluvial drainage areas (FDAs), and coastal drainage areas (CDAs). WBID centroids were joined to the CAF so that all stations in WBIDs were also mapped to an EDA, FDA, or CDA. Stations that were identified in the IWR database as "Stream" or "Blackwater" and had corresponding TN and TP data were identified and aggregated by EDA, FDA, or CDA. This resulted in approximately 145,000 paired observations or sampling events distributed in watershed stream networks across Florida. T he distributional statistics of all observations were computed for each watershed (Table 1-1).

The result of these analyses, and considerations based on FDEP bioregions, resulted in four NWRs that EPA proposed (U.S. EPA 2010a) (Figure 1-5). Following the proposal and a public comment period, EPA reconsidered the westward extent of the Hawthorn Group into the eastern portion of the proposed Panhandle Region and its influence on stream chemistry there (Figure 1-6). As noted above, when classifying Florida's streams, EPA identified geographic areas of the State having phosphorus-rich soils and geology, such as the northeastern part of Florida (i.e., the northern Apalachee River watershed and the northern Suwannee River watershed), and the area to the east of Tampa Bay. These areas are classified as separate NWRs (i.e., North Central and Bone Valley in Figure 1-5) because the naturally phosphorus-rich soils in these areas significantly influence stream phosphorus concentrations in these watersheds. EPA would expect from a general ecological standpoint that the associated aquatic life uses, under these naturally-occurring, nutrient-rich conditions, would be supported.

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Table 1-1. Distributional statistics (mean, median, 75th and 90th percentiles, and sample count) of stream TP and TN by estuarine watershed and aggregated by EPA NWR.

		Stream TP Concentration (mg/L)					Stream TN Concentration (mg/L)					Nutrient Watershed Region
EDA Code	Watershed Name	Mean	Median	75th %tile	90th %tile	Count	Mean	Median	75th %tile	90th %tile	Count	
G140x	Perdido Bay EDA	0.114	0.026	0.122	0.360	1219	1.55	0.71	1.61	4.35	1219	Panhandle West
G130x	Pensacola Bay EDA	0.030	0.020	0.035	0.054	2436	0.77	0.56	0.78	1.23	2436	Panhandle West
G120x	Choctawhatchee Bay EDA/FDA	0.040	0.027	0.045	0.066	1087	0.52	0.49	0.65	0.81	1087	Panhandle West
G110x	St. Andrew Bay EDA	0.019	0.011	0.020	0.020	678	0.50	0.36	0.55	0.75	678	Panhandle West
G100x	Apalachicola Bay EDA/FDA	0.047	0.027	0.047	0.077	1219	0.92	0.75	1.01	1.58	1219	Panhandle West
G090x	Apalachee Bay EDA	0.105	0.048	0.094	0.156	7523	0.88	0.63	0.94	1.45	7523	Panhandle East
G086x	Econfina-Steinhatchee CDA	0.091	0.069	0.104	0.165	777	0.91	0.87	1.17	1.41	777	Panhandle East
G080x	Suwannee River EDA/FDA	0.302	0.130	0.211	0.449	14469	1.13	1.03	1.29	1.62	14469	North Central
G070x	Tampa Bay EDA	0.595	0.290	0.620	1.270	14773	1.39	1.13	1.56	2.18	14773	West Central/ Peninsula
G060x	Sarasota Bay EDA	0.293	0.250	0.370	0.510	356	1.22	1.16	1.42	1.66	356	Peninsula
G050w	Charlotte Harbor EDA	0.526	0.350	0.638	1.051	6210	1.58	1.28	1.80	2.55	6210	Peninsula
S190x	Indian River/St. Lucie EDA	0.238	0.207	0.311	0.440	17182	1.45	1.37	1.66	2.03	17182	Peninsula
S183x	Daytona St. Augustine CDA	0.117	0.088	0.130	0.190	2190	0.98	0.94	1.22	1.52	2190	Peninsula
S180x	St. Johns River EDA/FDA	0.182	0.092	0.171	0.392	41428	1.51	1.22	1.67	2.44	41428	Peninsula
S175x	Nassau CDA	0.204	0.130	0.194	0.275	455	1.30	1.21	1.51	1.81	455	Peninsula
S170x	St. Marys River EDA	0.064	0.042	0.060	0.080	530	1.14	1.09	1.35	1.64	530	Peninsula
G078x	Waccasassa CDA	0.074	0.064	0.080	0.100	379	0.90	0.72	0.99	1.44	379	Peninsula
G076x	Withlacoochee River CDA	0.066	0.042	0.077	0.130	2555	1.09	0.97	1.30	1.73	2555	Peninsula
G074x	Crystal-Pithlachascotee CDA	0.076	0.031	0.090	0.180	4401	0.77	0.61	1.05	1.49	4401	Peninsula
G050a	Caloosahatchee River EDA	0.114	0.080	0.133	0.220	5630	1.05	1.00	1.40	1.71	5630	Peninsula
F100x	Kissimmee River/Lake Okeechobee FDA	0.311	0.175	0.440	0.706	19139	1.70	1.53	1.96	2.50	19139	Peninsula

EDA – Estuarine Drainage Area, CDA – Coastal Drainage Area, FDA – Fluvial Drainage Area.

Source: data obtained through Florida's IWR database.

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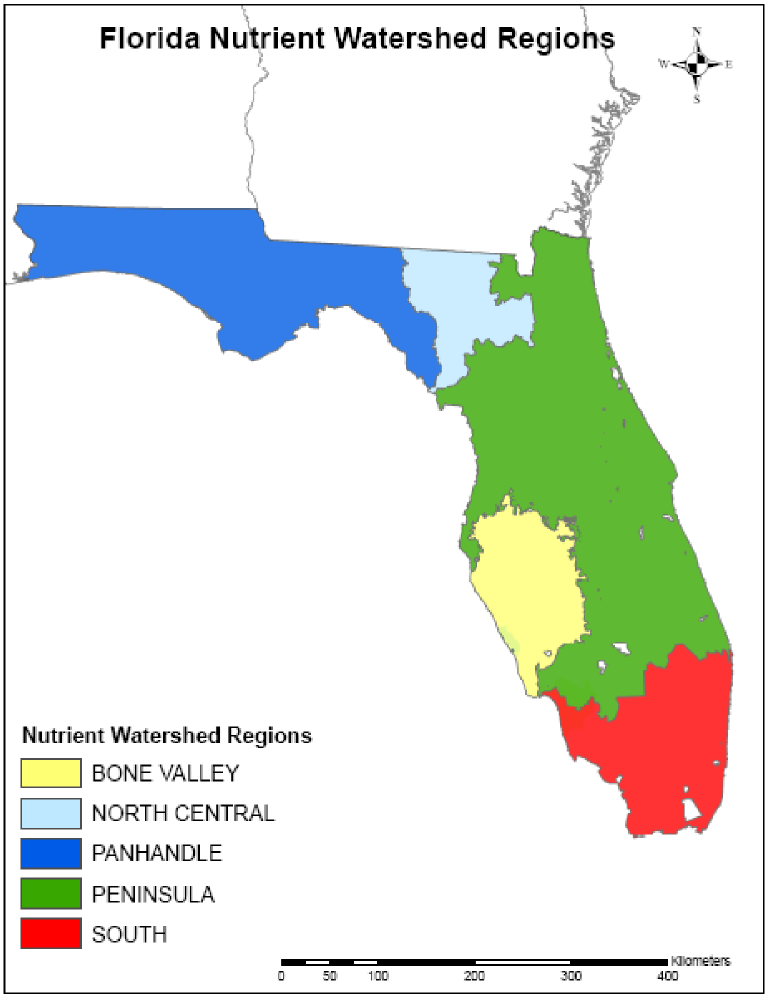

Figure 1-5. EPA's proposed Nutrient Watershed Regions (NWRs) for classifying Florida streams (U.S. EPA 2010a).

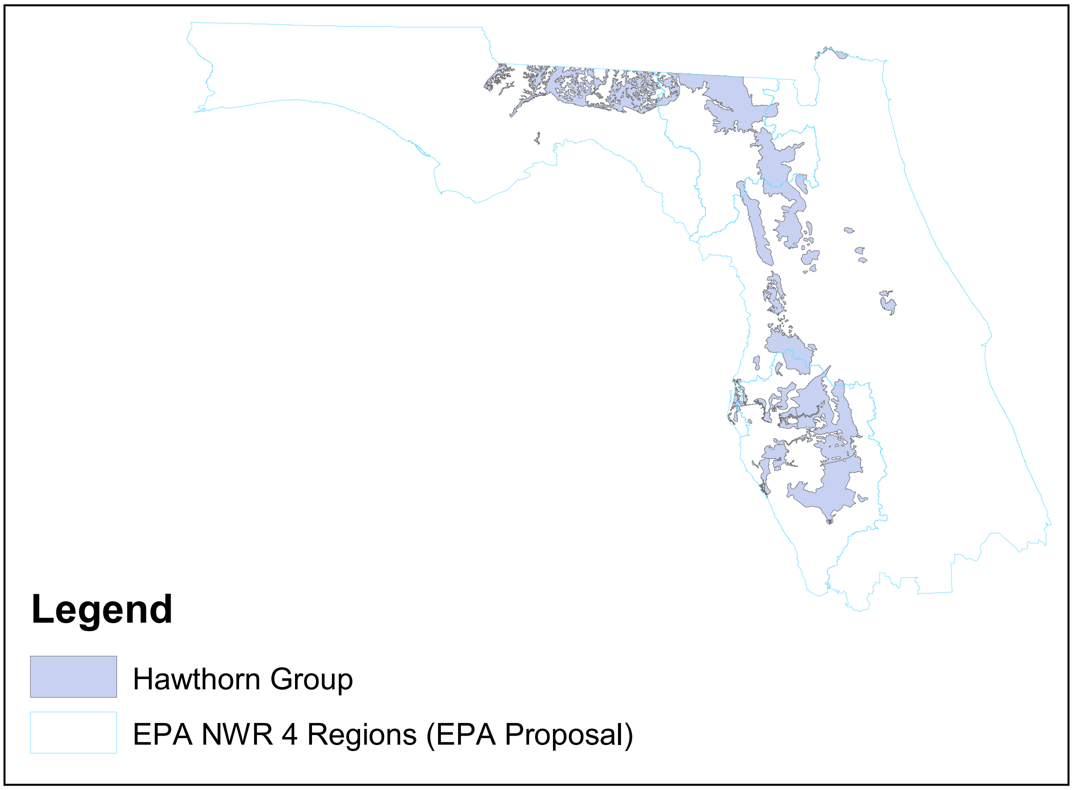

Figure 1-6. Map of EPA's proposed NWRs (U.S. EPA 2010a) and the Hawthorn Group indicating the presence of phosphate in the eastern Panhandle as well as the portions of the West Central (named the Bone Valley at proposal)

EPA revisited its exploration of underlying geological detail in the Panhandle and its relationship to observed patterns in stream chemistry. EPA took into account the portion of the Hawthorn Group that lies in the eastern portion of the Panhandle Region and explored delineation of the Panhandle Region along watershed boundaries into east and west regions. It appeared that higher TP concentrations were consistently associated with least-impacted streams in the eastern part of the Panhandle and this pattern could be explained by the underlying geology. EPA explored how well such a revised regionalization explained observed variability in TP concentrations relative to the proposed regionalization. EPA used a linear regression model to compare the variance in TP concentration explained by a four region model versus that explained by splitting the Panhandle into an east and west region along the Apalachicola River basin watershed boundary. Using either Benchmark Population or SCI Population approach, splitting the Panhandle Region into east and west regions explained more variability in TP concentrations than the original four stream bioregion model Table 1-2).

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Table 1-2. Results of linear regression model that modeled TP concentrations by different Nutrient Watershed Regions (NWRs). Regression models used the Benchmark Population and SCI Population.

Reference Population	Regionalization	R^2
Benchmark – TP	EPA's Proposed NWRs	0.52
Benchmark – TP	Revised NWRs with a divided Panhandle NWR	0.61
SCI – TP	EPA's Proposed NWRs	0.47
SCI – TP	Revised NWRs with a divided Panhandle NWR	0.53
Reference Population	**Regionalization**	R^2
Benchmark – TN	EPA's Proposed NWRs	0.52
Benchmark – TN	Revised NWRs with a divided Panhandle NWR	0.51
SCI – TN	EPA's Proposed NWRs	0.18
SCI – TN	Revised NWRs with a divided Panhandle NWR	0.15

The results in Table 1-2 above show that the regional refinement of dividing the Panhandle NWR into two regions provides a better representation of the natural range of nutrient conditions within the State, taking into account more of the natural variability, as measured by the linear regression models. This led EPA to conclude that a revision was necessary to divide the proposed Panhandle Region into two new regions—the Panhandle East, delineated at the western edge by the Apalachicola River watershed, and at the eastern edge by the Suwannee River watershed (or North Central NWR). EPA refers to this region as the Panhandle East and has effectively reduced in size the proposed Panhandle Region resulting in a Panhandle West NWR Figure 1-7, Figure 1-8). EPA has continued to adhere to watershed boundaries in its regionalizations to reflect the importance of the watershed context in nutrient source, fate, and transport. Smaller portions of Hawthorn Group cut across various smaller Peninsula drainages, but did not appear to contribute to sufficient systematic differences in TP concentrations in the receiving waterbodies to warrant further exploration or refinement.

EPA also re-evaluated its delineation of the West Central (referred to as the Bone Valley in EPA's proposed rule) and the Peninsula NWR. EPA found that its delineations of the West Central NWR was too broad and incorporated watersheds that were not influenced by underlying Hawthorn Group geology—especially coastal watersheds along the western boundary (Figure 1-9). EPA found that the Hawthorn Group boundaries are more constrained in this area than believed at its proposal (U.S. EPA 2010a) and that Charlotte Harbor and Sarasota Bay watersheds should not be included in the West Central for this reason. Therefore, EPA refined the boundaries for both the West Central and Peninsula NWRs. The result for the West Central NWR was a modified western boundary that shifts the Charlotte Harbor and Sarasota Bay drainages from the West Central (Bone Valley) to the Peninsula Region, because these two watersheds had minimal presence of Hawthorn Group elements at the land the surface (Figure 1-10). The Hillsborough River, eastern Tampa Bay drainages, Peace and Myakka River, and Sarasota Bay drainages are retained in the West Central NWR. These adjustments to the West Central NWR boundary (which also results in modifications to the Peninsula NWR) more accurately reflect the watershed boundaries and better reflect natural differences in expected stream chemistry. EPA provided these modifications subsequent to its proposed rule in the Agency's Supplemental notice published on August 3, 2010 (U.S. EPA 2010b).

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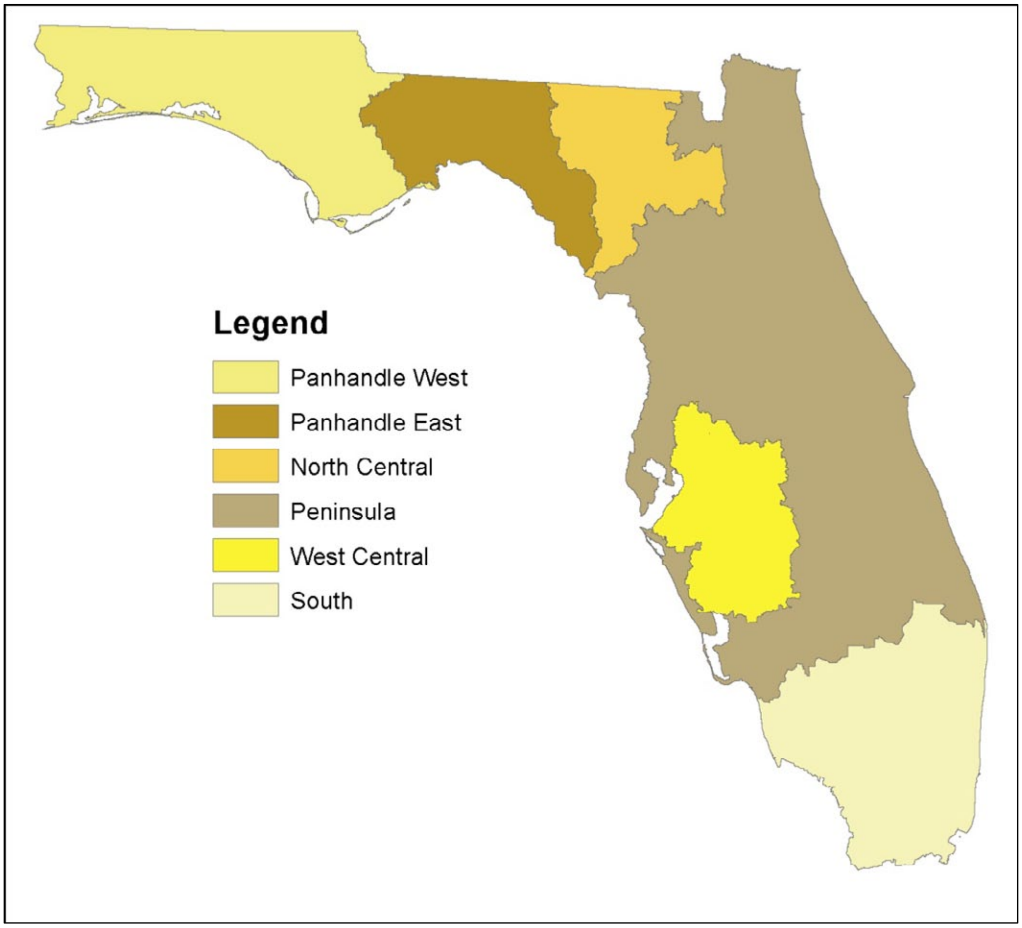

Figure 1-7. Map of EPA's stream classification by NWRs used in final rule.

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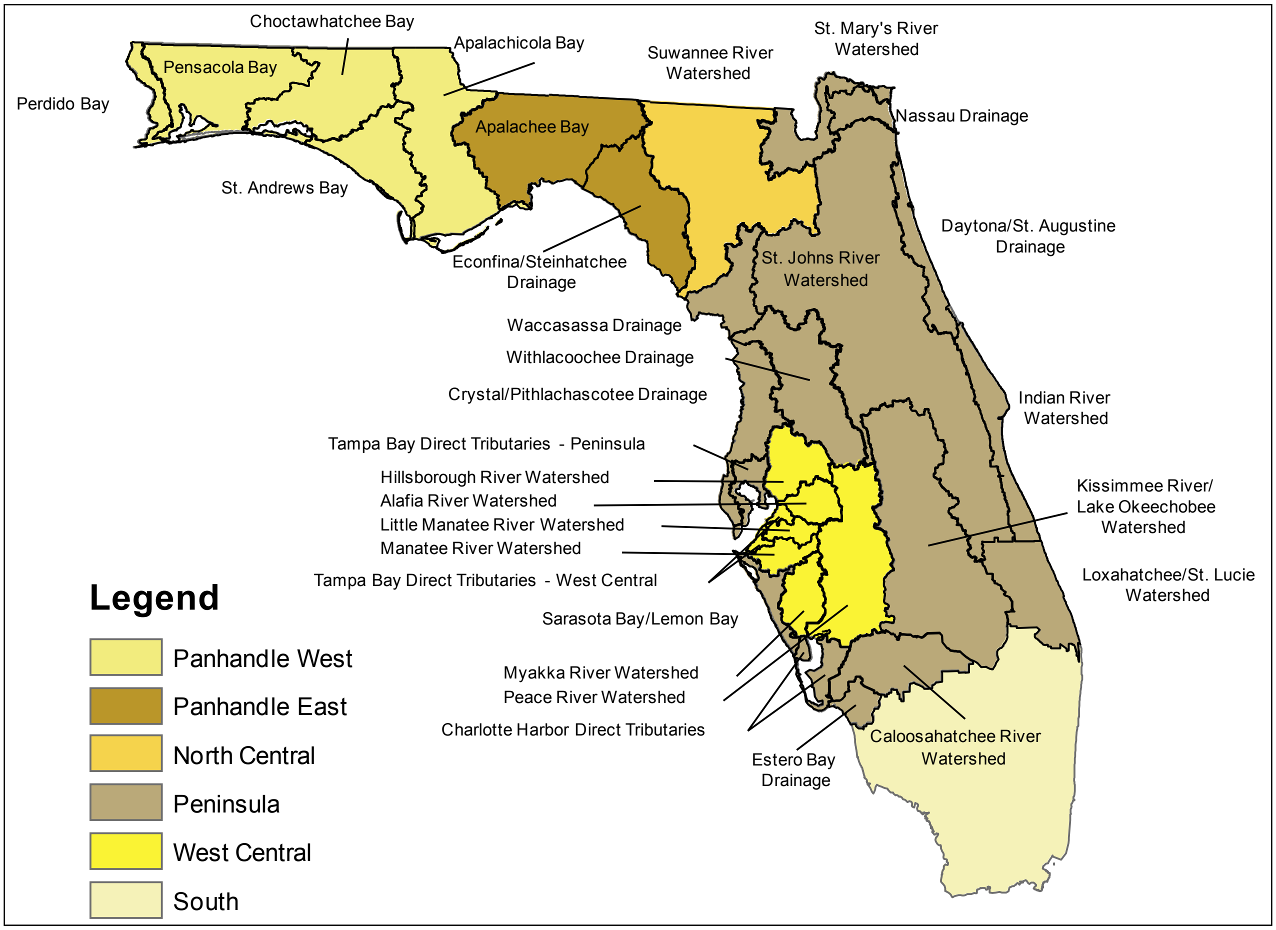

Figure 1-8. Detailed map of EPA's stream classification by NWRs used in final rule. Note that watershed boundaries are delineated by National Oceanic and Atmospheric Administration's (NOAA) Coastal Assessment Framework (CAF) of estuarine drainage areas (EDAs), fluvial drainage areas (FDAs), and coastal drainage areas (CDAs).

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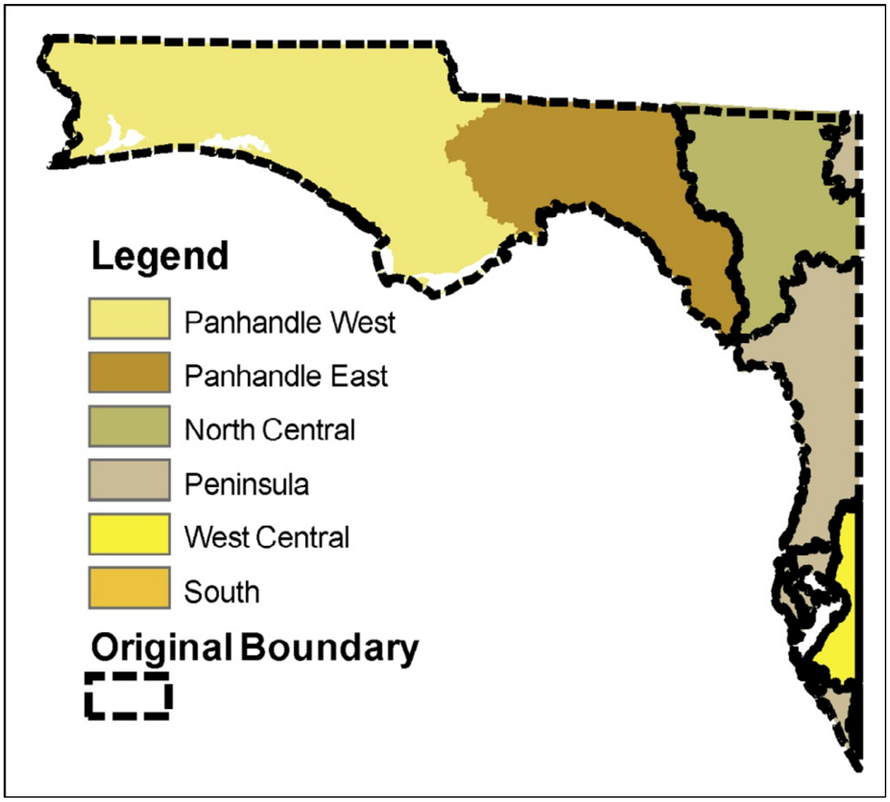

Figure 1-9. Detailed map of EPA's proposed Panhandle NWR boundary (U.S. EPA 2010a) and final Panhandle West and East NWRs.
Note that the new boundary between the Panhandle West and East is defined by the Apalachicola River watershed.

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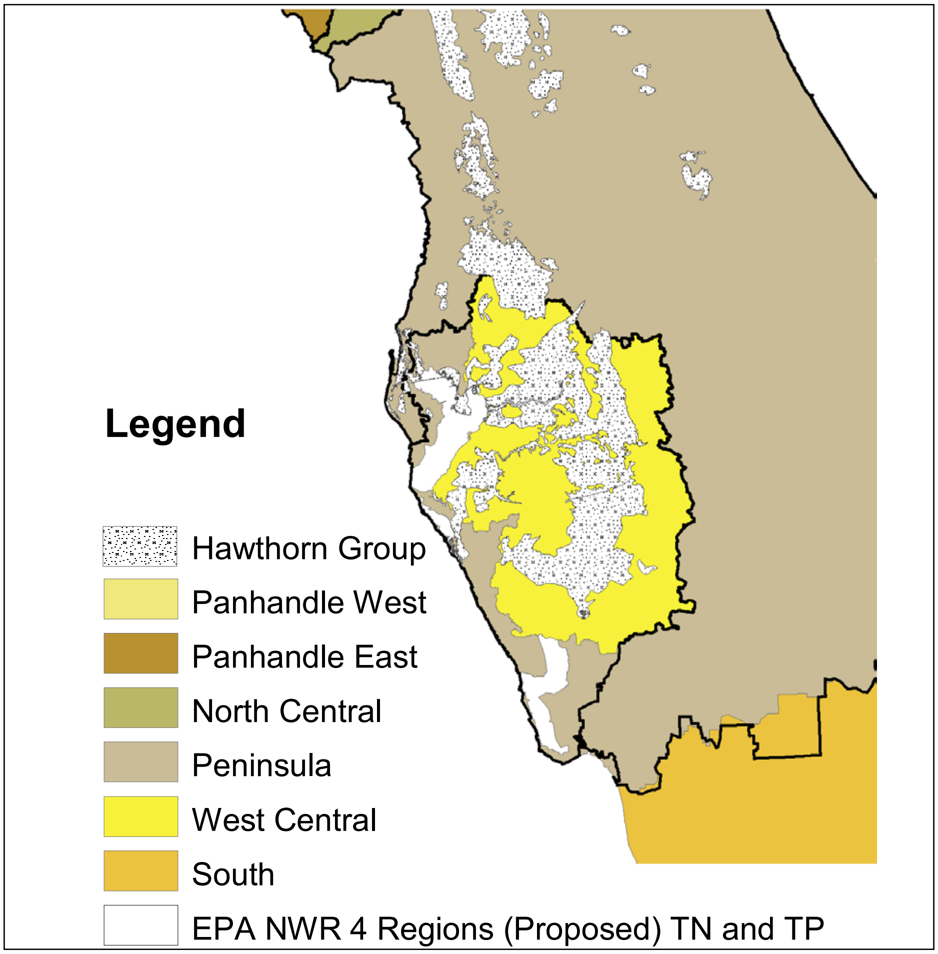

Figure 1-10. Map of the West Central NWR indicating the Hawthorn Group, EPA's proposed NWR boundaries (U.S. EPA 2010a), and EPA's final NWR boundaries.

EPA also made minor modifications to the map of the southern boundary of the Peninsula NWR to more accurately reflect the intended watershed boundaries proposed by EPA (U.S. EPA 2010a). The watershed boundary of the Caloosahatchee west of Lake Okeechobee and to the St. Lucie, Loxahatachee to the east defined this boundary. EPA also moved one WBID out of the map of the proposed Peninsula NWR because it was an inter-basin drainage that was not specifically part of the Caloosahatchee. EPA moved another WBID out of the map of the proposed Peninsula NWR because it was a coastal drainage to West Palm Beach and not to the St. Lucie/Loxahatchee Region. These WBIDs are in the South Florida Region, for which EPA will propose numeric nutrient criteria for flowing waters in 2011. EPA made additional minor modifications in the map of the southern boundary of the Peninsula NWR to ensure that WBID and drainage boundaries are concordant (Figure 1-11). A list of FDEP WBIDs organized by the NWR to which they belong is provided in *Appendix A1. Florida Waterbody Identification Numbers (WBIDs) by EPA Nutrient Watershed Region (NWR)*.

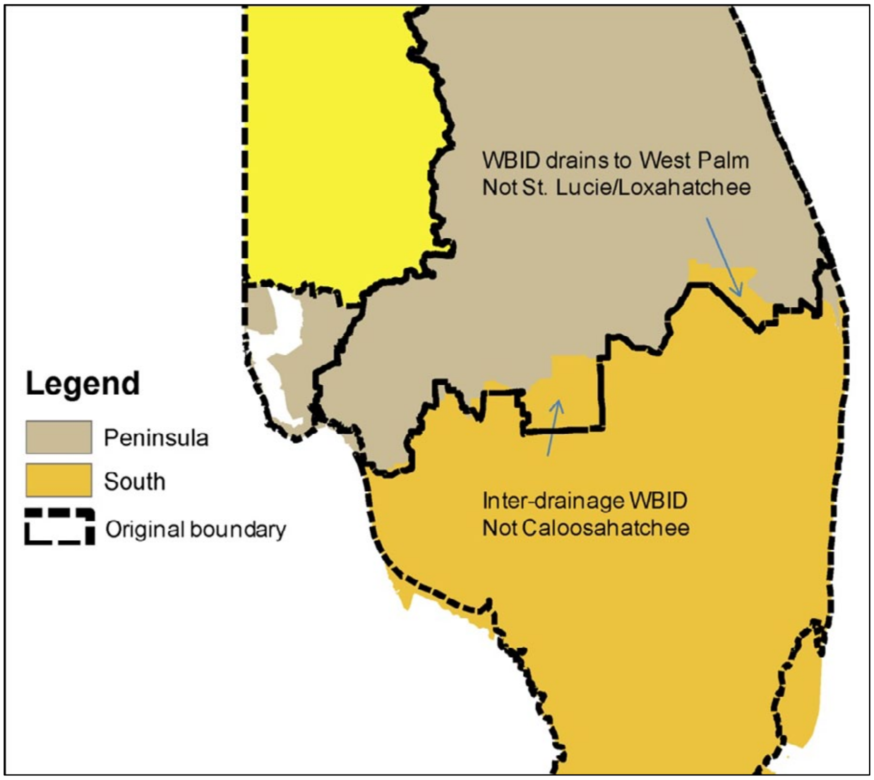

Figure 1-11. Map of the final Peninsula and South Florida NWRs and the original proposed NWR boundary.
Note that two of largest WBIDs that were moved from EPA's proposed Peninsula NWR to the South Florida NWR are indicated.

EPA chose to use the same regionalization for TN criteria as it used for TP criteria. EPA recognizes that the original and revised regionalization relied heavily on underlying geological differences among regions associated strongly with differences in phosphorus geology. These differences resulted in spatial differences in expected in-stream TP concentrations. At the same time, in-stream TN concentrations across the State were relatively more homogeneous across these same regions as compared to TP concentrations. EPA considered alternative approaches to classifying streams for purposes of TN criteria development. For example, EPA considered regionalizing streams and deriving numeric criteria based on a relationship between TN and stream color. EPA determined that setting criteria based on color assumes that the relationship between color and TN solely reflects the natural dynamics between the two parameters. EPA was concerned that anthropogenic effects on either color or nitrogen concentrations associated with runoff or effluent confound both the relationship between color and TN. EPA could not sufficiently address these concerns with the data available. As a result, EPA chose not to pursue a color based TN criterion at this time. However, EPA is open to such an approach for site-specific criteria derivation where a color-TN relationship can be affirmed as demonstrably independent from anthropogenic influence. EPA also considered regionalizing more broadly across two regions, similar to the approach considered by FDEP (2009), rather than the four regions described at EPA's proposal. Panhandle TN concentrations are lower than non-Panhandle Regions as evidenced by Figure 1-15 through Figure 1-18, and this is reflected in the resulting criteria.

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The statistical modeling indicates that a five region model explains no more or less variability than a two region model; this indicates that either approach is defensible. As a result, for consistency in application of criteria and in deference to the co-limiting nature of TN and TP, EPA has decided that similar regionalization for TN and TP is both defensible and protective of designated uses.

1.3 The Reference Condition Approach for Deriving Instream Protective TN and TP Concentrations

EPA used the reference condition approach for deriving instream protective TN and TP criteria. This approach, recommended and detailed in EPA guidance (e.g., U.S. EPA 2000), derives criteria from the distribution of values in a reference population, defined as "relatively undisturbed stream segments that can serve as examples of the natural biological integrity of a region" (U.S. EPA 2000, p. 94). For its final rule, EPA explored two reference populations by screening stream sites with nutrients data: (1) the Benchmark Population identified by screening for stream sites with nutrient data across the State of Florida based on land cover, aerial surveys, site visits, nitrate concentration, CWA section 303(d) impairments, and biological condition measures; and (2) the SCI Population, identified by screening stream sites with nutrient data based on macroinvertebrate samples on biological condition and CWA section 303(d) impairments. The final TN and TP criteria for each NWR are based on the Benchmark Population for all NWRs except the West Central Region where the criteria are based on the SCI Population approach detailed in EPA's proposed rule.

1.3.1 Data Selection and Preparation

The Florida Department of Environmental Protection extracted all available nutrient data from Florida's STORET (STOrage and RETreval) database and FDEP's Status and Trends database (Generalized Water Information System, GWIS). However, FDEP used only data from the STORET and GWIS databases submitted by water quality programs that met data quality standards, as required by FDEP QA rule 62-160 and their "Process for Determining Data Useability" (see *Appendix A2. FDEP's Process for Assessing Data Usability*). This process included screening all data points for potential data quality issues[3], such as improper sample preservation and analysis performed outside hold time. Data deemed to meet FDEP data quality requirements were extracted from thousands of sites statewide and is referred to as the *All Streams Data Set* (see *Appendix A3. Data Supporting EPA's Reference Approach for Deriving Numeric Nutrient Criteria for Florida Streams*). This initial population was then screened to identify reference condition sites for use in deriving nutrient criteria using the Benchmark and SCI Population approach described in Sections 1.3.2 and 1.3.3 below.

[3] EPA reviewed FDEP's data quality assurance procedures (FDEP 2008) that FDEP used to procure its data and determined that these procedures were consistent with EPA quality assurance policies.

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For the SCI Population analysis, only sites from the *All Streams Data Set* with macroinvertebrate data and multi-metric scores (SCI) were included. These SCI score data for Florida streams were obtained from FDEP in two spreadsheets—one with data from 2004 and before, and a second with data from 2004 through 2007. These spreadsheets contained SCI scores for specific sites across the State and additional data about the sites. They also contained associated water chemical and habitat data, including nutrient concentrations from grab samples taken when biological sampling occurred (hereafter referred to as grab sample chemistry). These two data sets were merged together into a final overall data set containing 2,023 samples from 1,115 sites within 614 WBIDs statewide (WBIDs are spatial units used in Florida for water quality management purposes). EPA based this analysis on WBIDs defined as of March 2010[4].

For both data sets, water chemistry data included TN, TP, and nitrate/nitrite (NO_3/NO_2) concentrations. TN was calculated by summing total Kjeldahl nitrogen (TKN-N) and nitrate/nitrite (NO_3/NO_2-N) concentrations from each sample. Water chemistry data were natural log (Ln) transformed prior to analysis to both adhere to assumptions regarding normality required of many parametric statistical tests and to reduce the influence of extreme values typical of log-normally distributed nutrient concentration data on estimates of central tendency. Figure 1-12 shows quantile plots of TP, Ln(TP), TN, and Ln(TN), indicating the non-normal distribution of raw TP and TN values and the improvement in the approximation of a normal distribution provided by natural log transformations.

Each site for which nutrient data were available was spatially linked to an NWR using a geographic information system (GIS) when the final NWR classification was determined. Because NWR boundaries adhered to WBIDs, both WBIDs and/or sites could be located within an NWR using the GIS. Final nutrient criteria were derived using the geometric mean of the annual average nutrient concentrations calculated for each site within a WBID. The geometric mean of the annual mean values is herein referred to as the "WBID average" concentration.

[4] EPA accessed FDEP's WBID GIS coverage in March 2010. This WBID coverage is provided in *Appendix A3. Data Supporting EPA's Reference Approach for Deriving Numeric Nutrient Criteria for Florida Streams*

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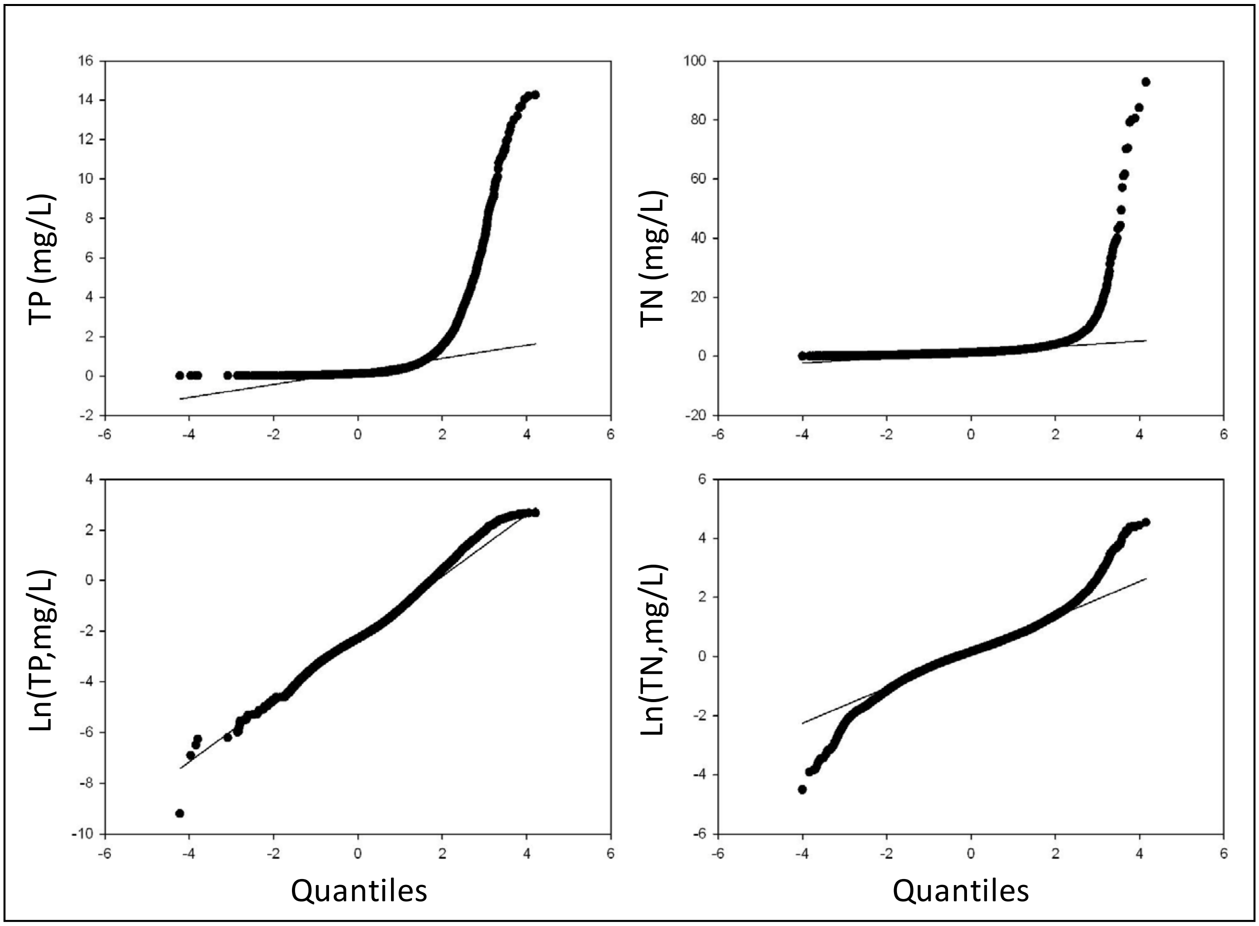

Figure 1-12. Quantile plots of TP, Ln(TP), TN, and Ln(TN).
Note that units are in mg/L.

1.3.2 Identification of Reference Population – Least Disturbed Sites (Benchmark Population)

For its final rule, EPA used a modification of an approach developed by FDEP, which identified a population of reference stream sites (Benchmark Population) that was least-disturbed by humans and nutrients. FDEP's approach was described in EPA's proposed rule (U.S. EPA 2010a); the modifications EPA was considering to that approach were described in the Agency's supplemental notice (U.S. EPA 2010b). FDEP developed a multi-step process (described below) to help ensure that the Benchmark Population selected represented the least human disturbance and provided designated use support. Similar to the SCI Population described in the proposed rule and below, streams documented to be least-disturbed by humans—excluding those with evidence of land disturbance, CWA section 303(d) impairments, and unhealthy biological conditions where known—can be used as part of reference condition approach because the range of nutrient concentrations observed at these sites support balanced natural populations of aquatic flora and fauna.

A critical component of EPA's modified benchmark approach was the multi-step evaluation process used to verify potential benchmark sites to ensure they represented least-disturbed

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conditions. This multi-step evaluation is outlined below. Screening to identify the Benchmark Population of least-disturbed reference sites began with the *All Streams* data set consisting of all available Florida nutrient data of known quality. Additional information about the verification process can be found in *Appendix A3. Data Supporting EPA's Reference Approach for Deriving Numeric Nutrient Criteria for Florida Streams*:

- The first screen was based on Florida's Landscape Development Intensity Index, which estimates the intensity of human land use. EPA removed potential benchmark sites with a corridor LDI score of greater than 2. The corridor LDI score quantifies land use within a 100 m corridor to each side and 10 km above candidate sites, using the method described in Brown and Vivas (2005). This step eliminated 76% of Florida sites from further consideration.
- Elimination of WBIDs included on the 2008 EPA-approved Florida CWA section 303(d) list of impaired waters based on failure to attain Florida's nutrient or dissolved oxygen water quality standards.
- Elimination of sites in WBIDs with mean nitrate concentrations greater than 0.35 mg/L, a concentration associated with excessive algae growth (see Section 3.3).
- Elimination of sites that, based on the professional judgment of FDEP scientists, did not represent minimally disturbed condition, considering the surrounding landscape conditions based on analysis of high-resolution aerial photographs taken in 2004–2005, input from FDEP district scientists knowledgeable of the area, and including FDEP watershed assessments, field visits, visual assessment of surrounding land use, and biological evaluations.
- Elimination of sites in WBIDs not meeting an average WBID SCI threshold of 40.
- Elimination of sites known to have near-field and/or watershed LDI scores > 3.

Maps, photos, and a summary of the data collected as part of this screening process at a subset of verified benchmark sites with nutrient concentrations greater than the mid-range of the distribution can be found in *Appendix A5. FDEP's Stream Benchmark Summaries*. Each of the steps outlined above is described in greater detail below.

a. Corridor LDI score of ≤ 2

Candidate benchmark sites were initially selected based on an application of the LDI. Brown and Vivas (2003) developed the LDI as an estimate of the intensity of human land use based on energy use per land use type. Application of the LDI is based on the ecological principle that the intensity of human-dominated land uses in a landscape affects the ecological processes of natural communities. More intense activities will result in greater effects on ecological processes. Natural landscapes with little or no agricultural or urban development have more intact natural ecological systems and processes. The LDI was developed specifically as an index of human disturbance, and it has been shown by Brown and Vivas (2003) to provide predictive capability regarding nutrient loading (Figure 1-13). LDI scores range from a high score of 10 that indicates high intensity land use to a low score of 1 that indicates natural lands.

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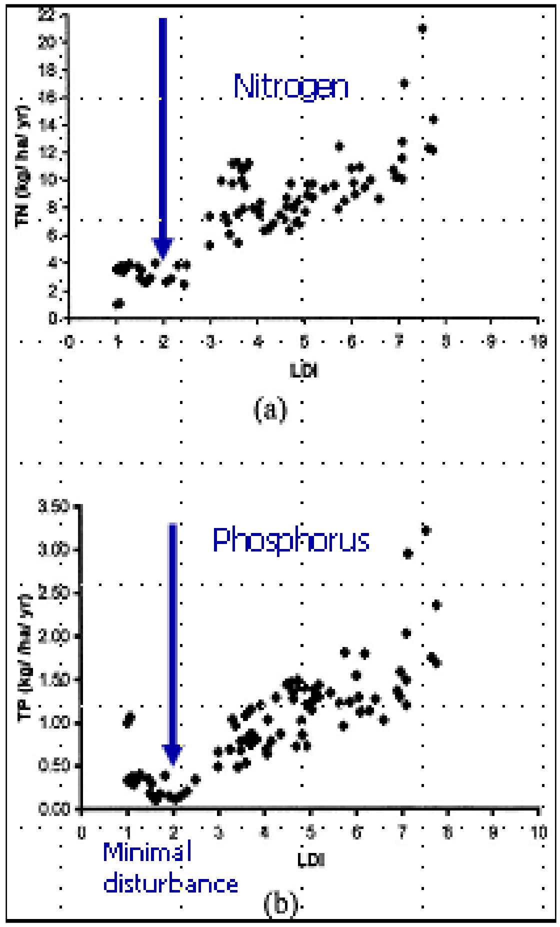

Figure 1-13. Relationship between nutrient loading, nitrogen in panel (a), phosphorus in panel (b), and the LDI in the St. Marks watershed, Florida (after Brown and Vivas 2003).
Note that the arrow is the LDI value at 2.0, which is the benchmark site threshold.

The LDI is calculated as the area-weighted value of land uses within an area of influence (Figure 1-14). Using land use coefficients and the percent area occupied by each land use as determined by GIS land use coverage developed from high-resolution aerial photographs, the LDI is calculated as follows:

$$LDI_{Total} = \sum (LDC_i \times \%LU_i)$$

where,

> LDI_{Total} = LDI for the area of influence
> $\%LU_i$ = percent of total area of influence in land use i
> LDC_i = LDI coefficient for land use i.

Sources of disturbance near a stream exert greater influence than do far-field human influences (Brown and Vivas 2003). In its benchmarks identification process, FDEP used the corridor LDI, calculated based on land uses within a 100 m corridor of a stream and extending 10 km upstream from a sample site (i.e., *"corridor approach"),* because this was found to be good predictor of ecological health (Fore 2004).

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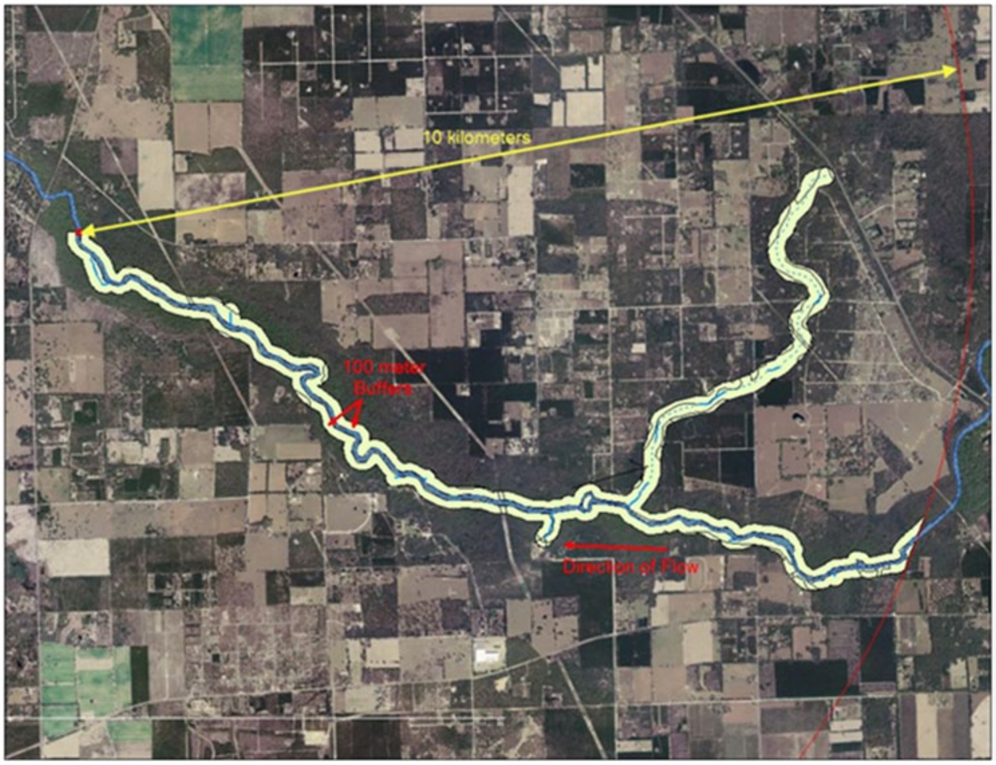

Source: FDEP (2009).

Figure 1-14. Depiction of land use area (light yellow arrow) included in an LDI calculation.

The utility of the corridor LDI approach is similar to the demonstrated effectiveness of riparian corridor zones in removing pollutants, especially nutrients, from stormwater inputs (both surface and subsurface flow). Studies have shown that corridor zone widths of 60 m are sufficient to reduce nutrient loads by up to 95% before reaching the stream (e.g., Peterjohn and Corell 1985). Additionally, Coastal Plain riparian corridors have been shown to be effective in retaining nutrients because of gradual slopes, permeable soils, and the abundance of roots that enter the shallow groundwater zones (Lowrance 1997). Because phosphorus can be bound to sediments, riparian zones retain the incoming phosphorus by capturing sediments. Other studies have shown that nitrate in shallow groundwater beneath riparian zones was removed by 85% to 90% due to plant uptake and denitrification in riparian zones 50 m to 70 m wide (Jacobs and Gilliam 1985; Jordan et al. 1993; Lowrance 1992, 1997).

For purposes of benchmark site selection, corridor LDI values were calculated from land uses within a corridor area of 100 m on each side of the stream and tributaries within a 10 km radius upstream of the sampling point as shown in Figure 1-14. While numerous studies have concluded that corridor widths of 50 m to 70 m are sufficient to reduce stormwater nutrient loads to streams by as much as 95%, additional corridor width can provide greater protection to the waterbody. Based on these literature findings and the better correlations with biological health described above, FDEP concluded that a corridor width of 100 m would provide indication of human-dominated land use and that an LDI calculated based on a 100 m corridor was an appropriate method of selecting candidate least-disturbed Benchmark Population reference sites.

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As discussed in FDEP's Nutrient Plan (2009), the LDI was specifically designed as a measure of human disturbance. LDI values of less than or equal to 2.0 within the 100 m corridor area are indicative of areas with less human impact. Other studies and evaluations have demonstrated, across other waterbody types and taxonomic groups, that the LDI is an accurate predictor of biological health; that is, healthy, well-balanced biological systems are much more likely to occur at sites with low LDIs (≤ 2.0) than at higher disturbance levels (Brown and Reiss 2006; FDEP 2009; Fore 2004; Fore et al. 2007; Niu 2004). The detailed methodology behind FDEP's corridor LDI approach is provided in *Appendix A4. FDEP's Method for Calculating a Corridor-based Landscape Development Intensity Index (LDI)*.

b. Screening against the 303(d) List of Impaired Waters

Sites located within WBIDs listed on the U.S. EPA-approved Florida 303(d) list as impaired for nutrients and/or dissolved oxygen were excluded as benchmark sites. EPA removed these sites and WBID from further analysis.

c. Screening against the 0.35 mg/L Proposed Nitrate+Nitrite Threshold

EPA screened data against its proposed nitrate+nitrite criterion of 0.35 mg/L, established for springs (U.S. EPA 2010a). This concentration is based on empirical stressor-response correlations from laboratory and field data. EPA determined for springs that concentrations of nitrate+nitrite of 0.35 mg/L or less are needed to prevent excess algal growth (see Chapter 3). Candidate benchmark sites that exceed this nitrate concentration (0.35 mg/L) were eliminated from the Benchmark Population of reference sites because the goal was to identify sites least-impacted by human disturbance. EPA deemed eliminating sites whose nitrate+nitrite concentrations are consistent with enrichment and established impacts was an appropriate screening step.

d. Elimination of Sites Based on Verification of Surrounding Land Use by Examining High-Resolution Aerial Photographs,Input from FDEP District Biologists, and Field Evaluation Including Watershed Assessment and Biological Appraisal of Benchmark Sites

Least-disturbed conditions of sites were confirmed through a review of recent (2004 to 2005) high-resolution (1 m ground resolution) aerial photographs by FDEP. That review consisted of searching the photos for recent land clearing or development, in particular any disturbance that encroached into the 100 m corridor area used to calculate the LDI, but also throughout the watershed. Additional sites were excluded based on FDEP's review of aerial photographs, including several sites that appeared to be tidally influenced, within canals, or artificially channelized. Therefore, these sites were not considered representative of a least-disturbed stream condition.

FDEP district scientists familiar with streams in their respective areas were asked to provide feedback on the list of candidate benchmark sites. Specifically, they were presented with the following information and question:

> "For ongoing nutrient criteria development, we are identifying sites with benign land uses in their upstream watershed (LDI < 2) to define the benchmark condition. Ken Weaver [of FDEP] has produced the attached table of low LDI peninsular benchmark sites. Can you please look over the list to determine if there are any human activities at particular sites,

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which may not have been captured by the LDI that would disqualify the site from being used to define *benchmark* for nutrient criteria?"

More than 20 study sites were excluded from the candidate benchmark population based on feedback and best professional judgment comments provided by FDEP district staff. The staff identified additional artificially channelized streams, estuarine sites, and potentially disturbed sites. In some cases, FDEP staff identified potential point source discharges or localized disturbances (e.g., cattle in the stream) that may not have been captured in the LDI calculation. In other cases, sites were excluded because the reviewer was aware of moderate to high levels of development within the watershed that while outside the 100 m corridor, in their opinion, could potentially increase nutrient concentrations.

Additional sites were excluded because they were potentially estuarine or tidally influenced based on proximity to the coast and a subsequent review of specific conductance data. All potentially estuarine sites routinely had specific conductance levels above 1,275 micro Siemens per cm (μS/cm) and episodic values above 4,500 μS/cm. A conductivity of 4,500 μS/cm is approximately equivalent to a chloride concentration of 1,500 mg/L, which is used in Florida as the threshold between predominantly fresh and marine waters and is also being used to define "predominately fresh waters" in the Agency's final rulemaking.

Results of the best professional judgment decisions based on aerial photographs and expert biologist opinion resulted in a screening score used to eliminate sites from the benchmark pool.

In 2007 and 2008, experienced FDEP staff conducted field verifications of a number of their candidate Benchmark Population sites, selected through the above process, as a means of providing additional assurance that sites were truly representative of least-disturbed conditions. Due to time and resource considerations, not all of their sites could be visited. Therefore, FDEP scientists selected candidate benchmark sites for additional review predominantly within WBIDs that had nutrient concentrations higher than the mid-range of their original distribution. The objective of this in-field verification step, which included a watershed survey and biological assessment, was to reinforce confidence in the selection of their final original Benchmark Population as representative of least-disturbed conditions. Representative sites within the target WBIDs were visited, and that site with the most extensive and longest period of nutrient data was selected to represent the WBID. Site evaluations included a survey of anthropogenic inputs and surrounding land uses. The survey included a driving tour of accessible portions of the watershed, guided by most recent high-resolution aerial photographs taken in 2004–2005 and maps of the drainage basin. During the watershed survey, FDEP investigators made a series of observations regarding potential human disturbances in the watershed, including potential nonpoint source inputs and hydrologic modifications (using the FDEP hydrologic scoring system). The hydrologic scoring system they used in their observations was originally developed to support the development of Florida's SCI. It is based on knowledge of water removal, patterns of drought, and hydrographs for the sites under evaluation, and it serves as a rough measure of hydrologic disturbance (Fore 2004, Fore et al. 2007).

Stream Habitat Assessments (HAs) were conducted by FDEP following standard operating procedures (DEP-SOP-001/01 FT 3100 (FDEP 2008)). The HA evaluates substrate condition and availability, water velocity, habitat smothering (e.g., by sand and silt), channelization, bank

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stability, and the width and condition of riparian vegetation. In addition to the 100 m reach of the stream examined during the HA, investigators also physically examined a minimum of 200 m upstream of the site, including potential riparian zone breaches.

At each site, trained and experienced FDEP scientific staff also collected and analyzed biological, chemical, and physical parameters (Table 1-3) following FDEP standard operation procedures (http://www.dep.state.fl.us/labs/sop). Water levels were evaluated both by reviewing hydrographs from the given stream or other streams in the general vicinity and by visually inspecting the stream habitats. Biological samples (e.g., SCI) were not collected if, based on the judgment of the experienced investigator, current or antecedent flow conditions were inappropriate, or a majority of the aquatic habitat was exposed to the air rather than being under water. Water chemistry samples were collected at all sites unless there were only discontinuous pools of water, in which case no samples were collected. These sites were, however, still included for consideration in the benchmark data set. Note that sites with an average SCI score of less than 40 on the SCI were excluded from the benchmark data set for calculation of the final nutrient distribution.

Table 1-3. Parameters monitored during the Benchmark Stream Survey.

Biological Parameters	Chemical and Physical Parameters
Stream Condition Index (SCI)	Total phosphorus
Rapid periphyton assessment	Nitrite + Nitrate
Qualitative Periphyton Sampling (i.e., periphyton taxonomy)	Total Kjeldahl Nitrogen
	Ammonia
	Color
Habitat assessment	Turbidity
Chl-*a*	Total suspended solids
Phaeophytin	Total organic carbon
Hydrologic modification scoring	Specific conductance (*in situ*)
Linear vegetation survey	Dissolved oxygen (*in situ*)
Percent canopy cover	pH (*in situ*)
	Water temperature (*in situ*)

Source: FDEP (2009).

Information acquired during the site and watershed evaluations was used to provide confirmation that benchmark sites were representative of least-disturbed conditions for the region and that nutrient concentrations support balanced natural populations of aquatic flora and fauna. The results of these surveys were incorporated into the best professional judgment (BPJ) scoring process described above.

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In addition to the screening steps used by FDEP to identify benchmark sites, EPA performed the following additional screening steps:

e. Elimination of WBIDS not meeting SCI threshold of 40

EPA calculated average SCI scores for each WBID and eliminated WBIDs with an average WBID SCI score of < 40. This screen was not available for all WBIDs because not all WBIDs had been sampled for macroinvertebrates. Thus, WBIDs without SCI scores were not eliminated. The development of the SCI and the significance of the threshold of 40 are discussed in more detail below and in greater detail in Section 1.4. As noted previously, the SCI is a bioassessment tool developed by FDEP and calibrated against the Biological Condition Gradient as an indication of the relative association of numerical SCI scores to ecological and biological attributes. Interval and equivalence tests and proportional odds logistic regression were used to discern a quantitative value above which Florida streams would be considered to contain healthy macroinvertebrate communities.

f. Elimination of sites with near-field and watershed LDI scores > 3

An additional analysis by EPA compared the nutrient distributions of sites characterized by the corridor LDI approach with LDI scores on the same sites calculated on a larger watershed area. EPA calculated near-field[5] and watershed[6] (or drainage basin) average LDI scores for each of the remaining 137 candidate benchmark WBIDs. For those WBIDs with average watershed or near-field LDI values above 3.0—the value that was correlated with a corridor LDI value of 2 and deemed to be equivalent to woodland pasture and unimproved pasture (Brown and Vivas 2005)—were also excluded from consideration. LDI scores were calculated using the same approach detailed above, but for the entire watershed and/or near-field region. That analysis suggested exclusion of two additional sites—one from the West Central NWR (South Prong Alafia River) and one from the North Central NWR (Camp Branch)—as well as closer scrutiny of additional sites in the West Central NWR. It indicated that higher TP concentrations observed at the South Prong Alafia River and Camp Branch could be explained by human disturbance, based on the higher watershed LDI scores found in the WBIDs. The analysis provided additional evidence that the benchmark reference site population, as identified by the corridor LDI approach, did not have extensive land use impacts beyond the 100 m/10 km scale examined by FDEP. In a general sense, EPA could not conclude from that analysis that a corridor LDI approach alone would yield a least-disturbed reference population with respect to nutrients. However, additional evaluations and analyses, such as those presented herein, could provide the quality assurance to support such a conclusion.

The initial set of candidate Benchmark Population reference sites identified statewide, with available data for TN or TP of known quality and LDI values ≤ 2.0, consisted of 1,512 sites distributed among 614 WBIDs. After excluding sites through the multi-step screening process, a total of 200 sites in 70 WBIDs remained. For ease of replication, a list of the screens is provided below for each step of the process after the initial LDI < 2 screen along with a step-by-step description of how EPA applied the screens to arrive at the final set of sites used in the criteria

[5] Near Field LDI values were computed from all drainage areas (at the 12-digit HUC level) in Florida within a 10 km straight-line radius to the most downstream site within a WBID.
[6] Watershed, or drainage basin, LDI values were computed from all drainage areas within Florida upstream of the most downstream site within a WBID.

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calculation (see *Appendix A3. Data Supporting EPA's Reference Approach for Deriving Numeric Nutrient Criteria for Florida Streams*).

1.3.3 Identification of Reference Population – Biologically Healthy Sites (SCI Population)

EPA proposed a reference population site selection process based principally on biological condition using the macroinvertebrate based stream condition index (SCI) developed by FDEP to evaluate biological condition in streams (U.S. EPA 2010a). That approach is utilized in the final rulemaking to derive criteria in the West Central Region. The data used to support this approach is detailed in *Appendix A3. Data Supporting EPA's Reference Approach for Deriving Numeric Nutrient Criteria for Florida Streams*. The process for identifying the biologically healthy SCI Population reference condition streams is described below.

a. Identifying biologically unhealthy site list

Sites were identified that did not meet the SCI threshold of 40 and were, therefore, not considered biologically healthy (see Section 1.4 for more on SCI evaluation). For the final rule, the average of SCI scores within a WBID was calculated and WBIDs with an average SCI score < 40 were excluded. The basis for the SCI threshold of 40 is provided in Section 1.4. The SCI is a bioassessment tool developed by FDEP and calibrated against the Biological Condition Gradient to interpret different SCI scores with respect to six distinct categories of biological condition. Interval and equivalence tests and proportional odds logistic regression were used to estimate an SCI value above which Florida streams would be considered to contain healthy macroinvertebrate communities (see Section 1.4.2).

b. Identifying impaired WBIDs

EPA excluded sites located within WBIDs listed on the U.S. EPA-approved Florida 303(d) list as impaired for nutrients and/or dissolved oxygen. EPA removed these sites and WBID from further analysis. A table of WBIDs listed as impaired for nutrients and/or dissolved oxygen was created and imported into a referential database. That database contained the nutrient chemical and SCI data described above, and individual sites within WBIDs listed for nutrient and/or dissolved oxygen impairments were identified and flagged.

c. Removing Sites

Any site flagged as biologically unhealthy (SCI < 40) or contained within a WBID listed for nutrients and/or dissolved oxygen impairment was removed from consideration as part of the biologically healthy site population for deriving the final instream protective criterion in the West Central Region. Once this screening selection was completed, EPA considered the remaining sites to be biologically healthy, with no evidence of impairment by nutrients and/or DO. Distributional statistics of TN and TP were then calculated for this biologically healthy stream population.

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1.3.4 Evaluation of Nutrient Concentrations and Distributions of Benchmark and SCI Populations within Each NWR

For illustrative purposes, maps of the distribution of TN and TP concentrations before and after screening steps (as described above in Sections 1.3.2) across NWRs for the Benchmark and SCI Populations (Figure 1-15 through Figure 1-22) are shown. Data are presented on a reference site and reference WBID scale. Cumulative percent frequency histograms for each NWR are also presented for TN and TP for the Benchmark and SCI Populations.

Figure 1-15 through Figure 1-22 show the degree to which the screening processes for the Benchmark and SCI Populations removed sites and WBIDs resulting in a smaller population of reference sites and WBIDs from which stream numeric nutrient criteria were derived.

Cumulative percent frequency histograms for TN (Figure 1-15B through Figure 1-18B) indicated that TN concentrations were relatively higher across the North Central, Peninsula, and West Central NWRs, compared to the Panhandle West and East NWRs, in both the Benchmark and SCI Populations, regardless of the scales (site or WBID). Low TN sites and WBIDs were distributed across the Panhandle West and East NWRs, with generally lower concentrations in the Panhandle West NWR. Cumulative percent frequency histograms for TP (Figure 1-19B through Figure 1-22B) indicated that TP concentrations were higher in the West Central and North Central NWRs, compared to the Panhandle West, Panhandle East, and Peninsula NWRs. Higher TP concentrations appear to localized in the northern half of the North Central NWR. Within the Panhandle and Peninsula NWRs, some high concentrations of TP at sites and WBIDs were observed, but there did not appear to be a consistent pattern.

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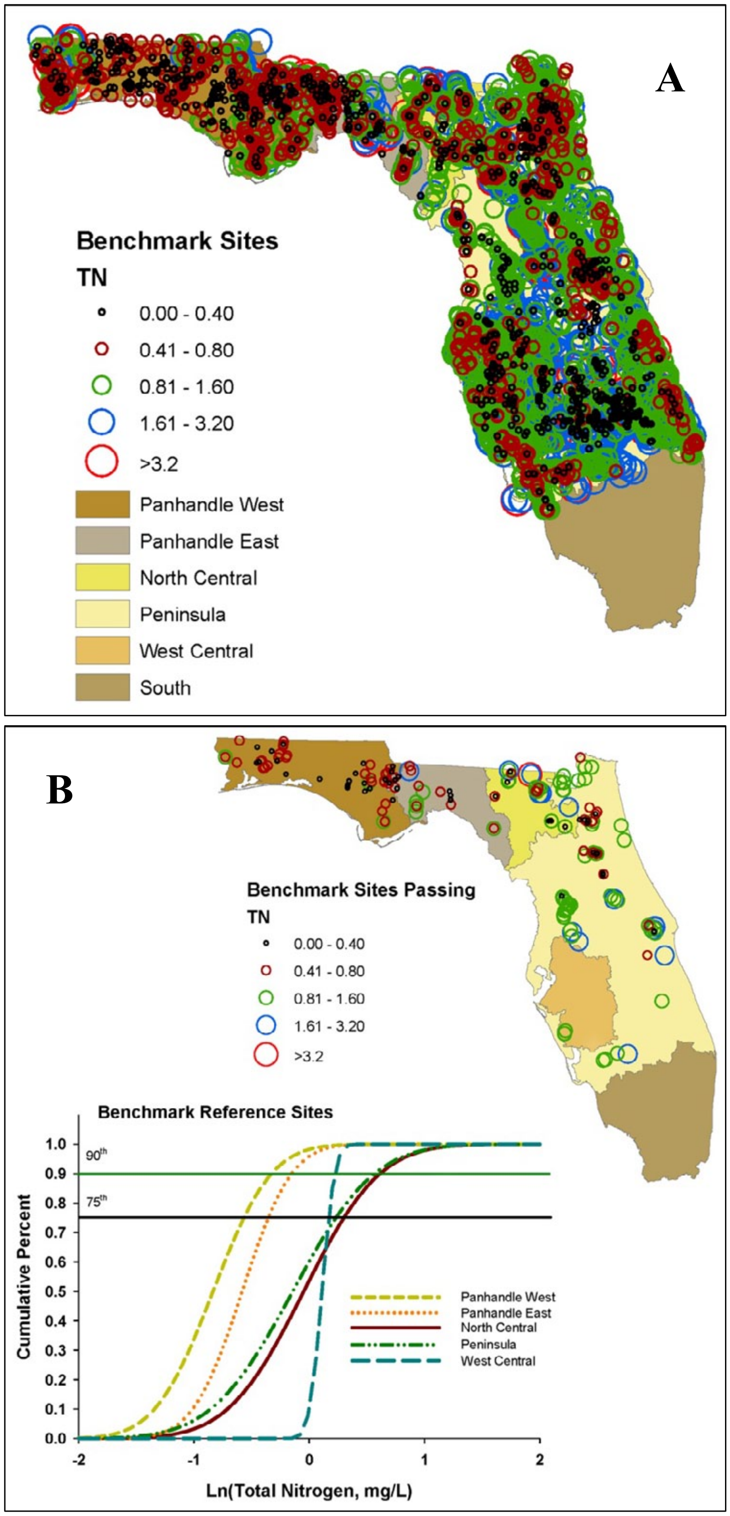

Figure 1-15. Plot of TN concentrations among benchmark reference sites across EPA's NWRs for the Benchmark Population before (A) and after (B) screening.
Note that concentrations graphed are Ln(TN, mg/L). Also shown (in B) are cumulative percent frequencies of TN concentrations at benchmark sites.

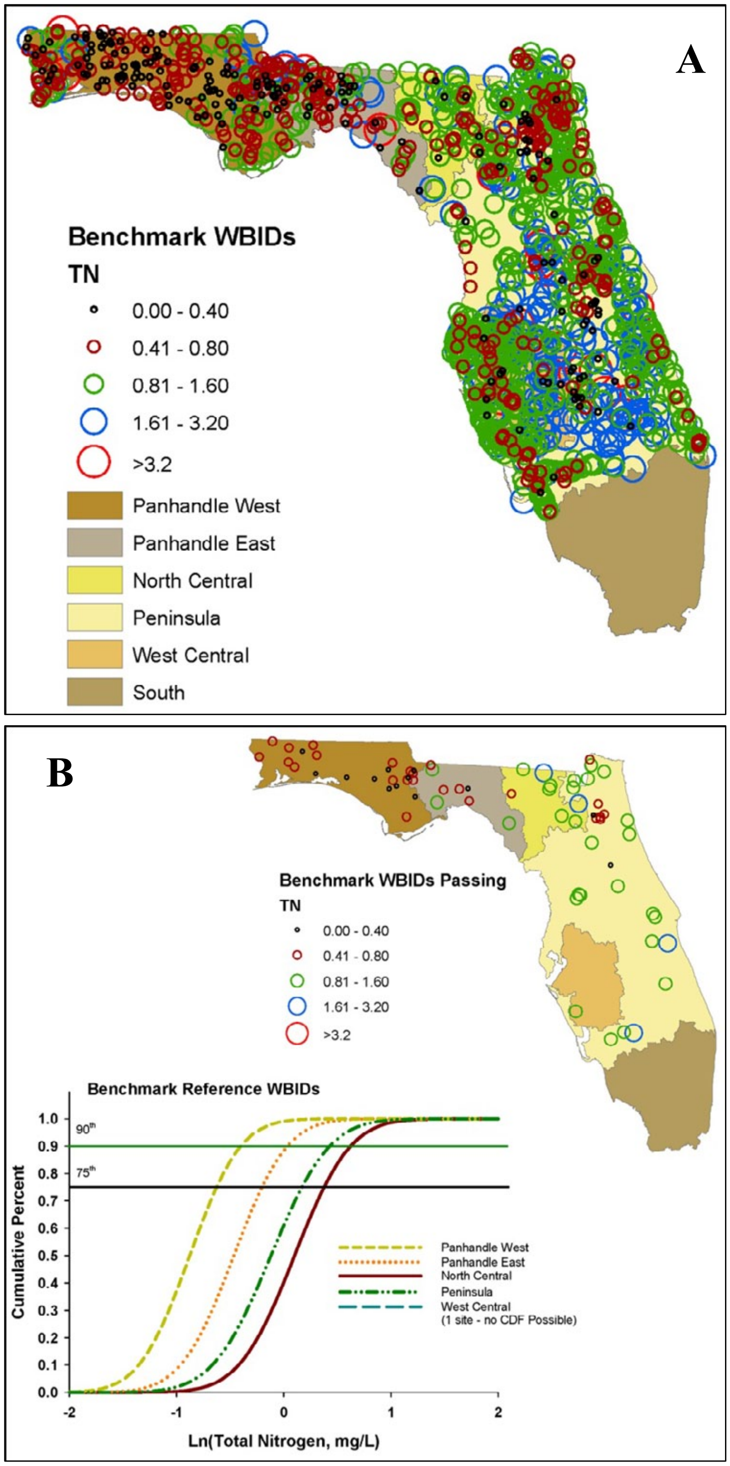

Figure 1-16. Plot of TN concentrations among benchmark reference WBIDs across EPA's NWRs for the Benchmark Population before (A) and after (B) screening.
Note that concentrations graphed are Ln(TN, mg/L). Also shown (in B) are cumulative percent frequencies of TN concentrations at benchmark WBIDs.

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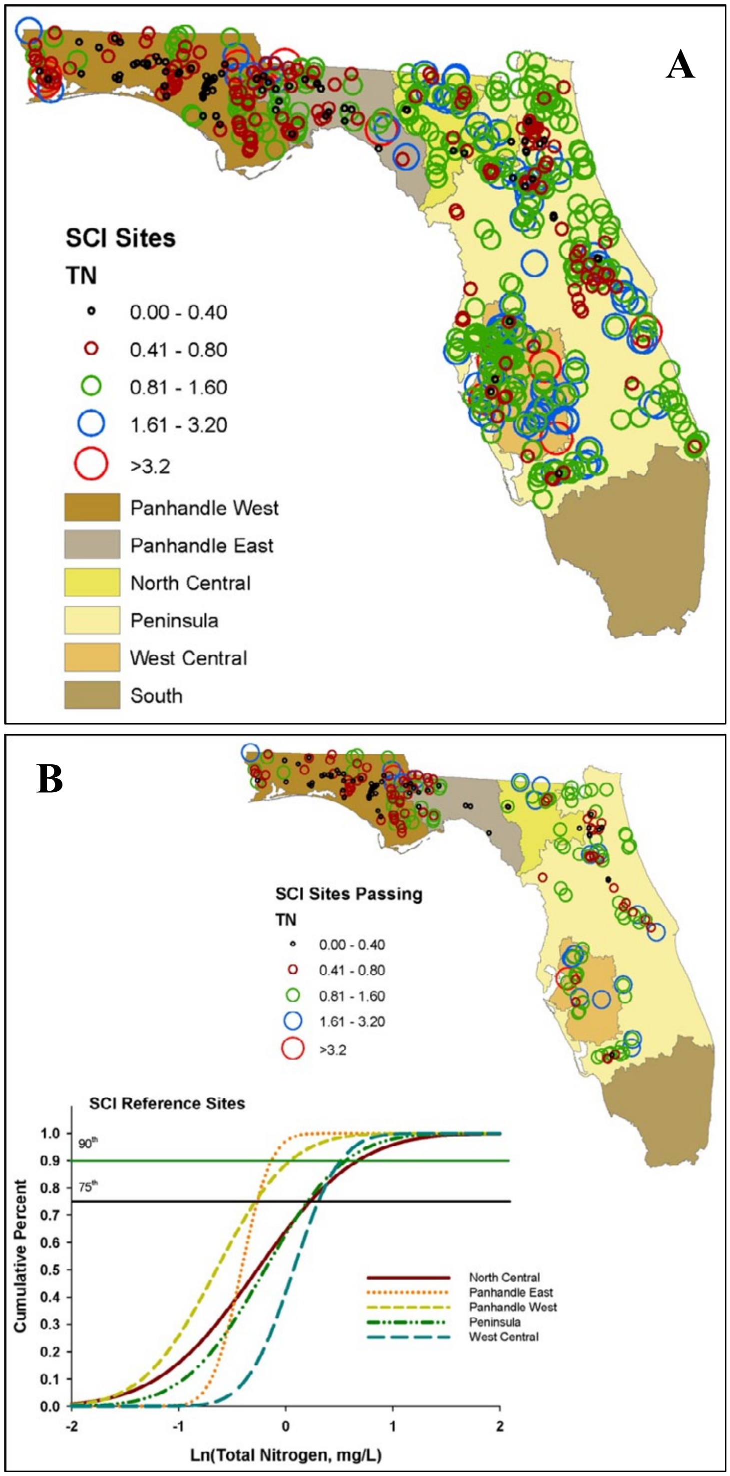

Figure 1-17. Plot of TN concentrations among SCI reference sites across EPA's NWRs for the SCI Population before (A) and after (B) screening.
Note that concentrations graphed are Ln(TN, mg/L). Also shown (in B) are cumulative percent frequencies of TN concentrations at SCI sites.

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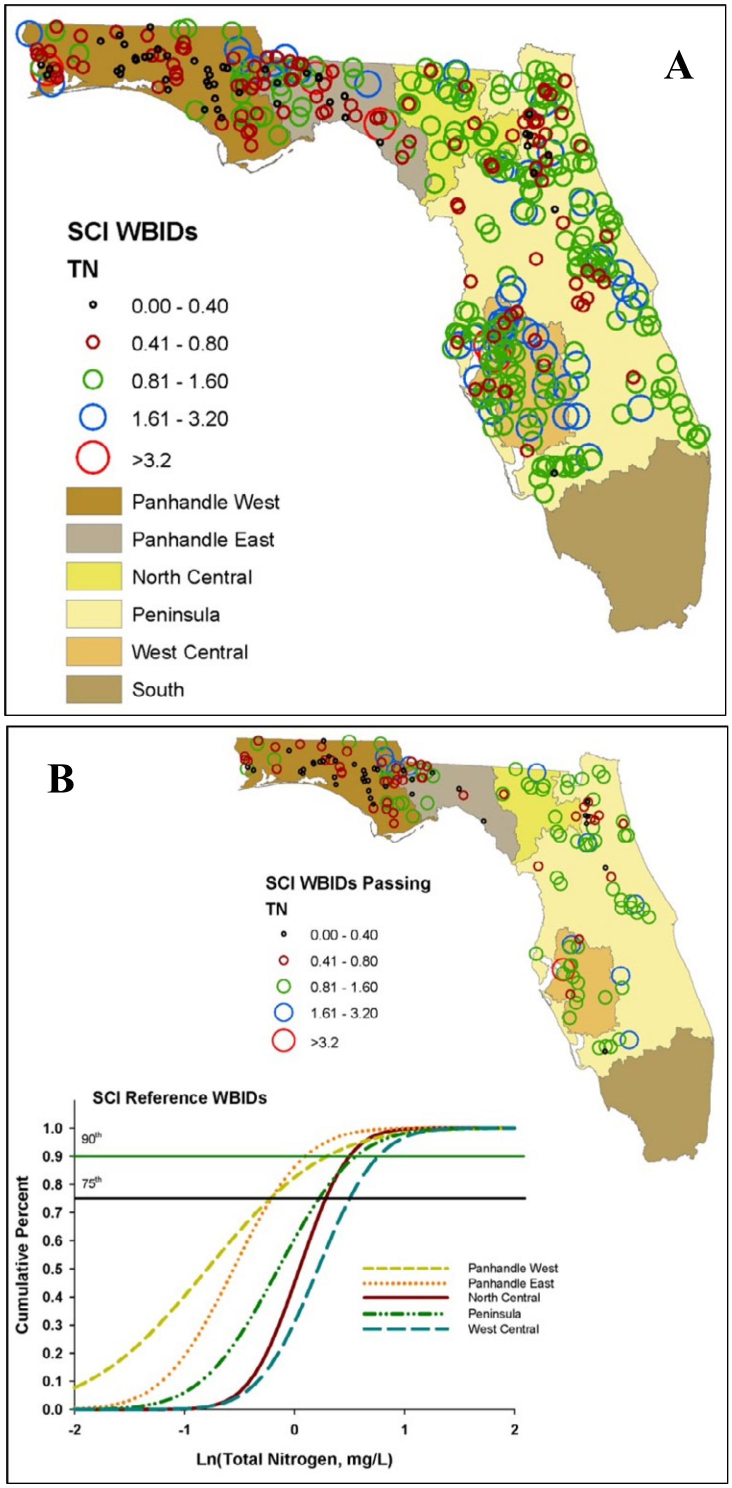

Figure 1-18. Plot of TN concentrations among SCI reference WBIDs across EPA's NWRs for the SCI Population before (A) and after (B) screening.
Note that concentrations graphed are Ln(TN, mg/L). Also shown (in B) are cumulative percent frequencies of TN concentrations at SCI WBIDs.

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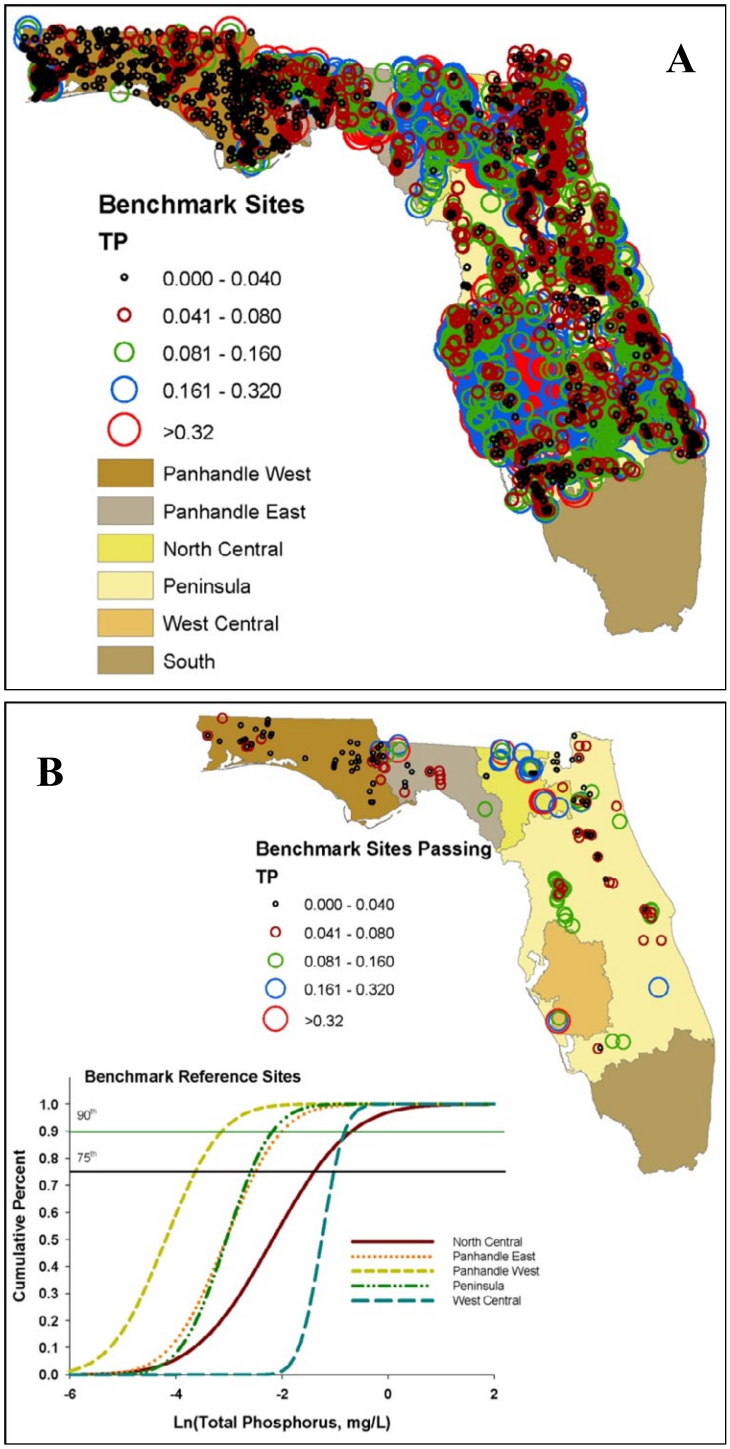

Figure 1-19. Plot of TP concentrations among benchmark reference sites across EPA's NWRs for the Benchmark Population before (A) and after (B) screening.

Note that concentrations graphed are Ln(TP, mg/L). Also shown (in B) are cumulative percent frequencies of TP concentrations at benchmark sites.

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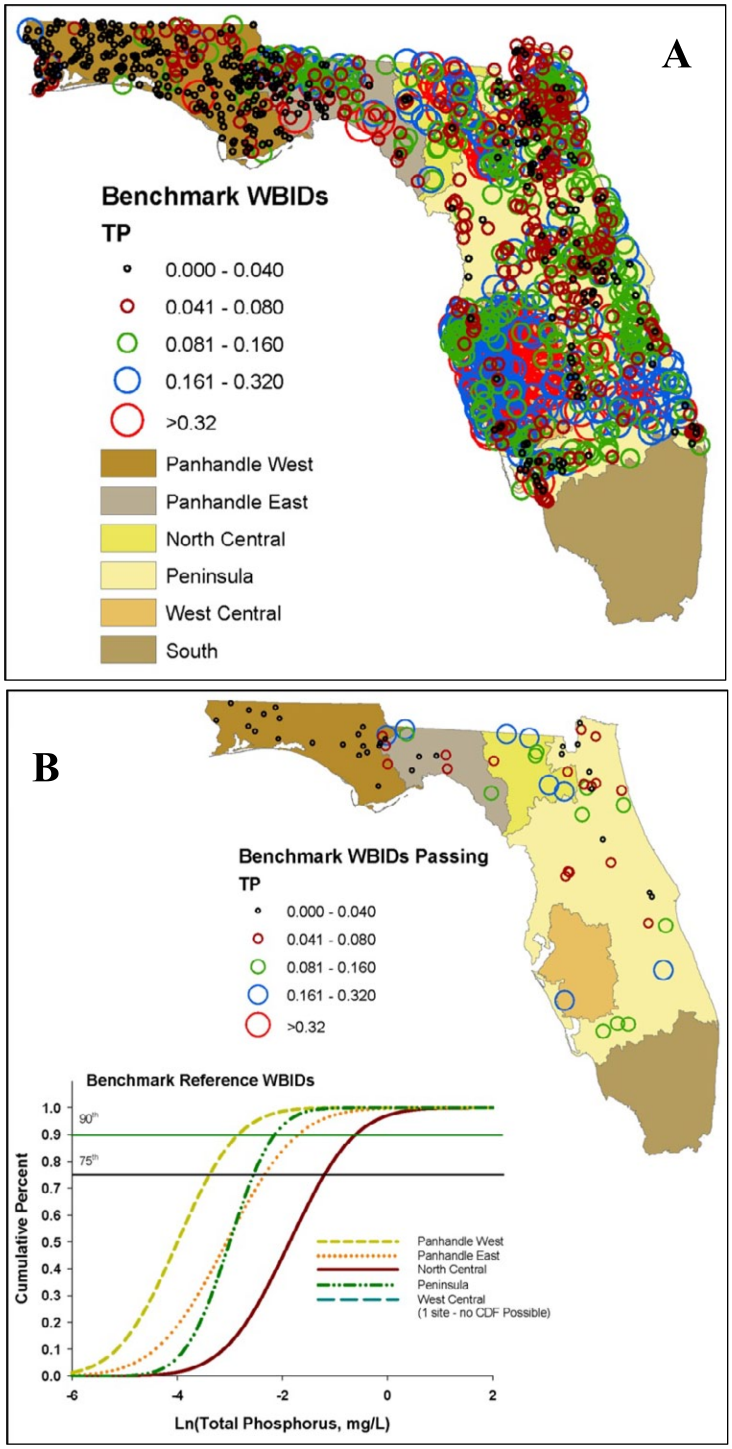

Figure 1-20. Plot of TP concentrations among benchmark reference WBIDs across EPA's NWRs for the Benchmark Population before (A) and after (B) screening.
Note that concentrations graphed are Ln(TP, mg/L). Also shown (in B) are cumulative percent frequencies of TP concentrations at benchmark WBIDs.

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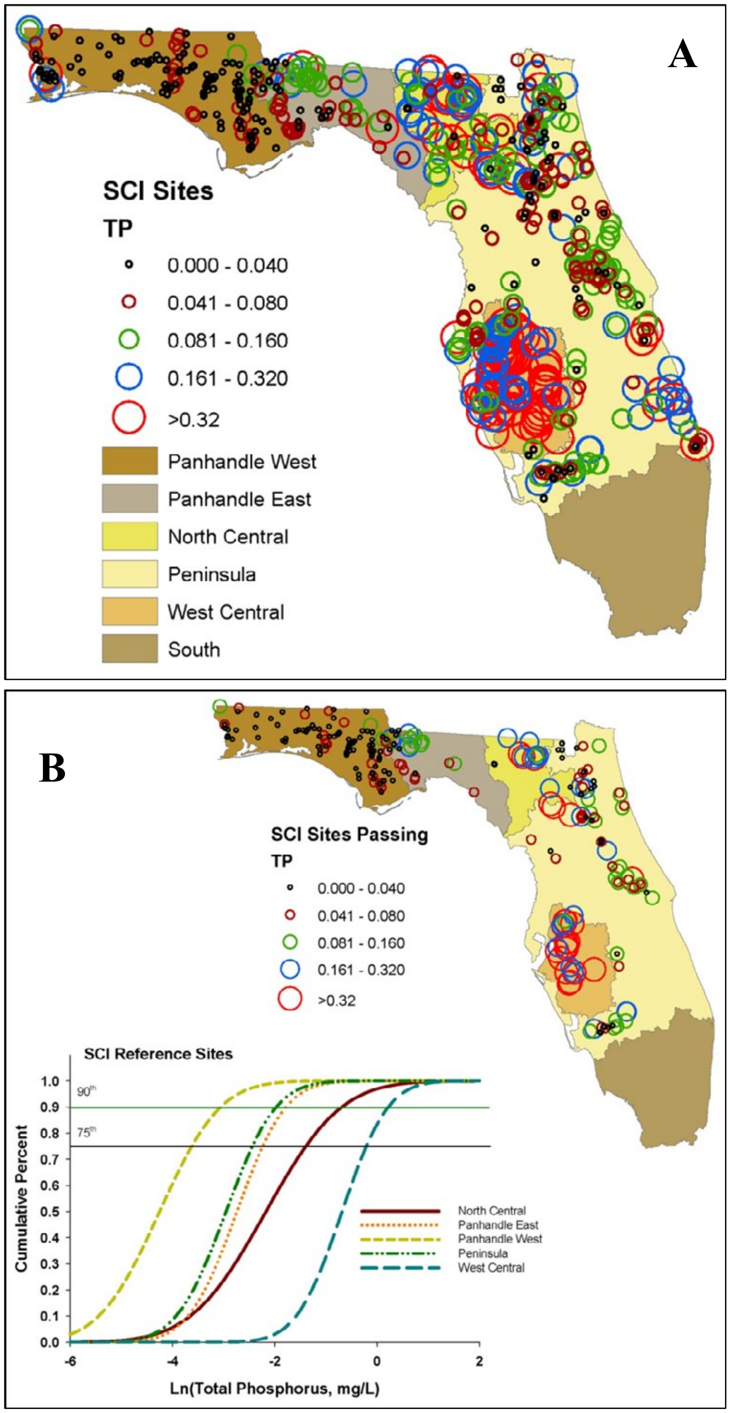

Figure 1-21. Plot of TP concentrations among SCI reference sites across EPA's NWRs for the SCI Population before (A) and after (B) screening.
Note that Concentrations graphed are Ln(TP, mg/L). Also shown (in B) are cumulative percent frequencies of TP concentrations at SCI sites.

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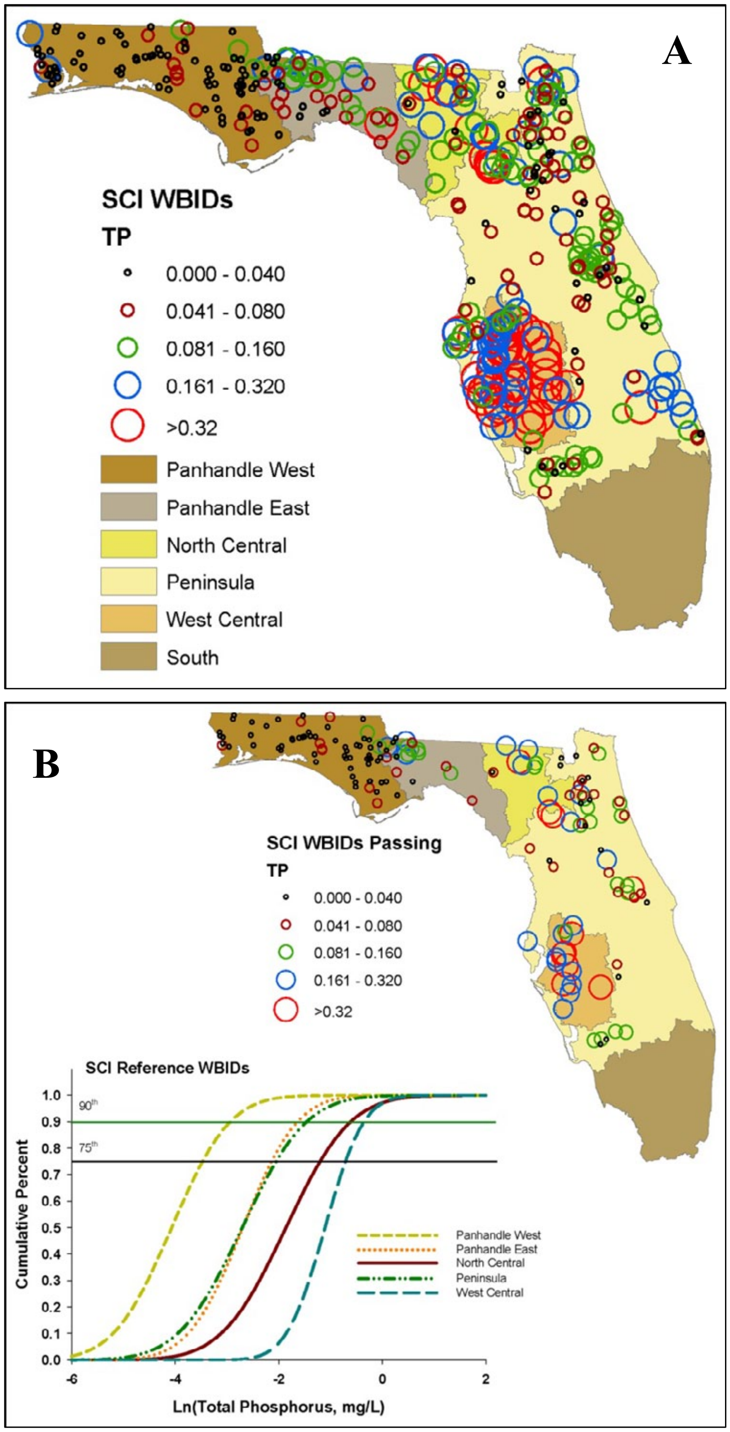

Figure 1-22. Plot of TP concentrations among SCI reference WBIDs across EPA's NWRs for the SCI Population before (A) and after (B) screening.

Note that concentrations graphed are Ln(TP, mg/L). Also shown (in B) are cumulative percent frequencies of TP concentrations at SCI WBIDs.

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1.3.5 Calculation of Numeric Nutrient Criteria – Benchmark and SCI Populations

EPA described in its proposal (U.S. EPA 2010a) and supplemental notice (U.S. EPA 2010b) the calculation of numeric nutrient criteria for TN and TP for Florida streams. EPA calculated TN and TP criteria by NWR using both the Benchmark and SCI Populations based on a WBID-year average and site average basis, respectively. These results are presented in Table 1-4.

Table 1-4. EPA's proposed numeric nutrient criteria for streams presented in the Supplemental notice (in bold).

EPA Benchmark Population – WBID-Year Averages						
Parameter	**Region**	**Mean (Ln, mg/L)**	**Standard Deviation (mg/L)**	**75th (mg/L)**	**90th (mg/L)**	**N**
TP (mg/L)	Panhandle West	-4.12	0.73	0.03	**0.04**	63
	Panhandle East	-2.96	0.58	0.08	**0.11**	42
	North Central	-1.96	0.72	0.23	**0.35**	79
	Peninsula	-3.07	0.65	0.07	**0.11**	75
	West Central	-1.39	0.51	**0.35**	0.48	8
TN (mg/L)	Panhandle West	-0.94	0.36	0.50	**0.62**	65
	Panhandle East	-0.67	0.50	0.72	**0.97**	38
	North Central	0.11	0.41	1.48	**1.90**	68
	Peninsula	-0.03	0.42	1.29	**1.67**	72
	West Central	0.12	0.21	**1.30**	1.47	3
EPA SCI Population – Site Averages						
Parameter	**Region**	**Mean (Ln, mg/L)**	**Standard Deviation (mg/L)**	**75th (mg/L)**	**90th (mg/L)**	**N**
TP (mg/L)	Panhandle West	-4.09	0.99	**0.03**	0.06	111
	Panhandle East	-2.82	0.80	**0.10**	0.17	26
	North Central	-1.84	1.22	**0.36**	0.76	17
	Peninsula	-2.87	0.98	**0.10**	0.20	86
	West Central	-0.77	0.68	**0.73**	1.11	22
TN (mg/L)	Panhandle West	-0.73	0.82	**0.84**	1.38	109
	Panhandle East	-0.61	0.52	**0.77**	1.06	25
	North Central	-0.10	0.72	**1.48**	2.29	17
	Peninsula	-0.24	0.64	**1.20**	1.77	85
	West Central	0.31	0.41	**1.80**	2.31	22

Note: the mean, standard deviation, calculated 75th and 90th percentiles, and sample size of TP and TN distributions are presented for each NWR. EPA proposed the 90th percentile of the Benchmark population (computed from WBID-years), with the exception of the West Central for which EPA proposed the 75th percentile. EPA also proposed the 75th percentile of the SCI population (computed from site averages).

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For the final rule, EPA calculated TN and TP criteria based on both the Benchmark and SCI Populations using a WBID average basis (Table 1-5). EPA considered a range of factors in its approach to deriving numeric nutrient criteria for Florida's streams in final rule, including the spatial and temporal variability of a WBID or site and time scales over which to average water quality data. EPA also considered the distributional statistics for each reference population and how they might influence the derivation of scientifically defensible numeric nutrient criteria. These factors are discussed below.

In general, the reference site approach for deriving nutrient criteria is based on the premise that nutrient concentrations at reference sites represent the natural variability of concentrations within a given study area. In Florida, Benchmark and SCI Populations approaches were used to establish the variability of long-term average nutrient concentrations across each NWR. EPA conducted analyses to identify a spatial averaging scheme (WBID vs. site) that best represented the spatial variability of nutrient concentration within the State, and to identify a temporal averaging scheme (long-term average vs. annual averages) that best represented the long-term average.

Spatial statistical analysis of geometric mean TP concentrations at different sites across Florida indicated that TP concentrations at sites that were close together tended to be more similar than sites that were distant from one another. This trend was identified by computing the semi-variance associated with different separation distances using the following equation:

$$\gamma(h) = \frac{1}{2n(h)} \sum_{i=1}^{n(h)} (\overline{TP}(x_i + h) - \overline{TP}(x_i))^2$$

Where $\gamma(h)$ is the semi-variance for a separation distance of h, $n(h)$ is the number of samples separated by h, x_i is the location of site i, and $TP(.)$ is the geometric mean concentration at the indicated location. Semi-variances increased with separation distances to a distance of approximately 35 km, which indicated that TP concentrations at sites separated by > 35 km in general were not associated with one another (Figure 1-23). Total phosphorus concentrations were more similar between adjacent sites as the separation distance between those sites decreased.

Because the average distance between each Benchmark Population site and its nearest neighbor was approximately 3.7 km, and so, TP concentrations at many sites in the Benchmark Population were associated with one another. Including all TP concentrations from all of these sites would, in essence, over-represent certain areas of Florida in which many sites were sampled in close proximity with one another. In contrast, the average distance between each WBID and its nearest neighbor was 19 km. Based on these results, EPA concluded that summarizing nutrient concentrations by WBID, rather than site-by-site, provided a distribution of nutrient concentrations that was more representative of the overall spatial variability of nutrient concentrations across the State. The same analysis was conducted for TN with similar results and conclusions.

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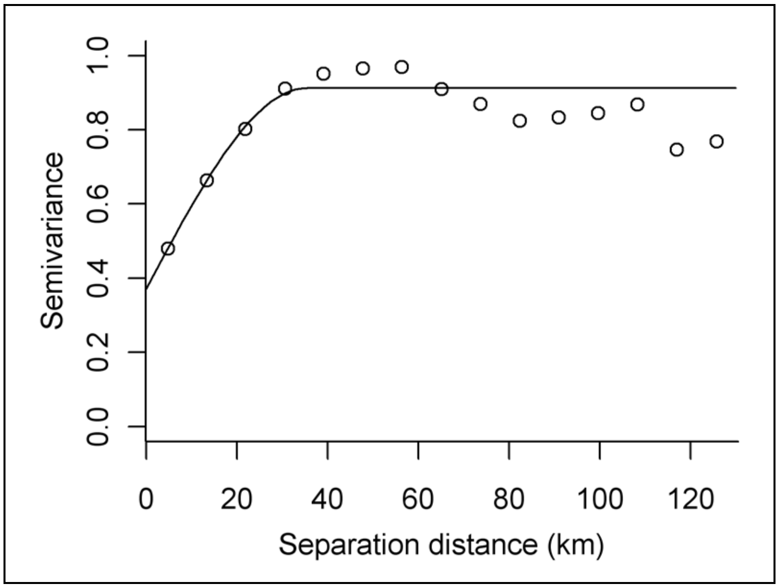

Figure 1-23. Semi-variogram showing the degree to which TP concentrations at closely located sites are auto-correlated.

Benchmark and SCI populations in Florida were intended to represent the spatial variability of long-term average background nutrient concentrations across the State. To this end, a single geometric mean value computed for each WBID was identified as the most appropriate temporal averaging scheme. The WBID-year averaging approach, in which nutrient concentrations at each WBID were summarized by annual geometric mean, and then included as separate values in the distribution, has the advantage of increasing the sample size used for estimating criteria. However, different annual averages from the same WBID do not necessarily vary in the same way as values observed across different WBIDs. The difference between the magnitude of temporal variability within a WBID and the magnitude of spatial variability across WBIDs can be seen qualitatively by comparing time series of TP concentrations in four Panhandle Benchmark WBIDs (Figure 1-24). The magnitude of variability within each WBID was much smaller within each site (i.e., the spread of values for any set of points with the same color) compared to the magnitude of variability across different sites.

Formal statistical analysis supports these qualitative insights. A linear mixed model was used to quantify the variance associated with across-site variability and within-site inter-annual variability. The results of this model indicated that the across-site variance was approximately 17× greater than within-site inter-annual variance for TP. For TN, the ratio of across-site variance to within-site inter-annual variance was approximately 10:1. Thus, including multiple annual averages from a given WBID as distinct values in the reference distribution over-represents that particular WBID. Based on these findings, EPA decided that using a single long-term average from each WBID provided a more representative distribution of nutrient concentrations for deriving criteria.

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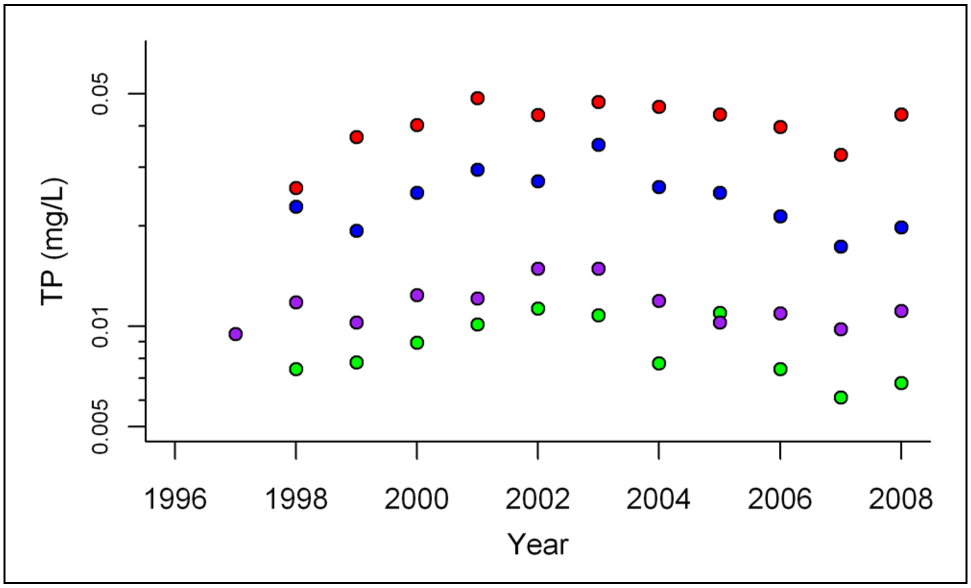

Figure 1-24. Time series of annual geometric mean TP concentrations at four Panhandle benchmark WBIDs.
Note that different colored symbols denote different WBIDs.

Once EPA organized TN and TP data by WBID averages, EPA calculated criteria as the 90th percentile of annual geometric mean concentrations in WBIDS from each NWR (Panhandle East and West, North Central, and Peninsula nutrient regions) using the following equation:

90th Percentile = exp[mean(Ln(nutrient concentration)) + 1.282 SD (Ln(nutrient concentration))]

The criterion for the West Central NWR was calculated as the 75th percentile annual geometric mean concentration based on the SCI Population. A lower percentile was selected for the West Central due to greater uncertainty in the upper end of the distribution for this region given the number of SCI sites and annual average values. In the West Central NWR, benchmark conditions were available for only one WBID (see Table 1-5). The equation for the 75th percentile is the following:

75th Percentile = exp [mean(Ln(nutrient concentration)) + 0.674 SD(Ln(nutrient concentration))

The selection of percentiles was based both on a judgment of reference site quality and statistical considerations. Selection of a central tendency of the reference distribution (i.e., the median or geometric mean of a log-normal distribution) would imply that approximately half of the reference sites are not attaining their uses. Alternatively, the upper end of the distribution (e.g., the 90th percentile) is appropriate if there is confidence that the distribution truly reflects minimally impacted reference conditions and can be shown to be supportive of designated uses (i.e., balanced natural populations of aquatic flora and fauna).

EPA concluded that the Benchmark Population and the resulting benchmark distributions of TN and TP were based on sufficient evidence of least-disturbed reference conditions after the additional quality assurance screens applied by EPA. Use of many different screens reduced the effects of uncertainty inherent in any single screening threshold. Thus, EPA found it reasonable to use the 90th percentile of WBID annual averages from this population to establish the final

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rule criteria in four of the five NWRs, which is consistent with EPA's previously published guidance on percentile selection (U.S. EPA 2000). Because of the limited number of Benchmark Population sites in the West Central, EPA relied upon the SCI Population to derive TN and TP criteria in this region. Fewer screens were used to identify the final SCI Population, and hence, identifying these sites as least-disturbed was more uncertain than the Benchmark Population approach. Furthermore, as described in detail above, the SCI approach does not rely on a quantitative assessment of potential human disturbance through the use of surrounding land cover analysis of stream corridor and watershed land development indices (i.e., LDI). For these reasons, EPA reasoned that the stream criteria in the West Central Region should be based on the 75^{th} percentile values of the distribution from the SCI Population, which is also consistent with EPA's previously published guidance (U.S. EPA 2000).

EPA explored calculating TN criteria for two NWRs as part of its consideration of a two region TN model (see above): the Panhandle NWR and combined Northeast, North Central, Peninsula and West Central NWR. Nitrogen concentrations did not exhibit the same regional spatial patterns as that observed for phosphorus. The primary regional difference was between the Panhandle and the remainder of the State. However, there was no loss of variance explained by a five NWR model over a two region mode (Table 1-2). This result combined with the potential for co-limitation of nutrients, the similarity of nitrogen concentrations within Panhandle and non-Panhandle Regions, and the practical benefits of consistent regionalization, EPA decided to use the same NWRs for TN as for TP.

In conclusion, EPA determined that the most appropriate approach for summarizing available nutrient concentration data in the Benchmark and SCI Populations was to calculate single long-term annual average values within each WBID. The distributional statistics of a WBID average approach are shown in Table 1-5. For comparative purposes, the results of a WBID average approach are compared to EPA's approaches WBID-year and site average approaches for the Benchmark and SCI Populations, respectively, described at proposal (U.S. EPA 2010a) and Supplemental notice (U.S. EPA 2010b). These comparisons are shown in Figure 1-25 and Figure 1-26, respectively.

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Table 1-5. EPA's final numeric nutrient criteria for streams (in bold).

EPA Benchmark Population – WBID Averages						
Parameter	**Region**	**Mean (Ln, mg/L)**	**Standard Deviation (mg/L)**	**75th (mg/L)**	**90th (mg/L)**	**N**
TP (mg/L)	Panhandle West	-3.992	0.900	0.03	**0.06**	22
	Panhandle East	-3.056	1.060	0.10	**0.18**	10
	North Central	-2.045	0.669	0.20	**0.30**	8
	Peninsula	-2.999	0.666	0.08	**0.12**	29
	West Central	-1.508	NC	NC	NC	1
TN (mg/L)	Panhandle West	-0.876	0.371	0.53	**0.67**	22
	Panhandle East	-0.464	0.388	0.82	**1.03**	10
	North Central	0.101	0.407	1.46	**1.87**	8
	Peninsula	-0.119	0.427	1.18	**1.54**	29
	West Central	0.189	NC	NC	NC	1
EPA SCI Population – WBID Averages						
Parameter	**Region**	**Mean (Ln, mg/L)**	**Standard Deviation (mg/L)**	**75th (mg/L)**	**90th (mg/L)**	**N**
TP (mg/L)	Panhandle West	-4.05	0.87	0.03	0.05	69
	Panhandle East	-2.68	0.80	0.12	0.19	17
	North Central	-2.00	1.16	0.30	0.60	11
	Peninsula	-2.69	0.98	0.13	0.24	50
	West Central	-1.11	0.59	**0.49**	0.70	15
TN (mg/L)	Panhandle West	-0.79	0.85	0.80	1.34	69
	Panhandle East	-0.55	0.51	0.81	1.10	17
	North Central	0.05	0.35	1.33	1.65	11
	Peninsula	-0.15	0.55	1.25	1.75	50
	West Central	0.21	0.43	**1.65**	2.14	15

Note: the mean, standard deviation, calculated 75th and 90th percentiles, and sample size of TP and TN distributions are presented for each NWR. EPA is finalizing the 90th percentile of the Benchmark Population (computed as a WBID average), with the exception of the West Central for which EPA is finalizing the 75th percentile of the SCI population (computed as a WBID average). Mean and standard deviation are in Ln(mg/L) and percentiles are in mg/L. NC – not computed, or descriptive statistics that could not be computed due to limited sample size.

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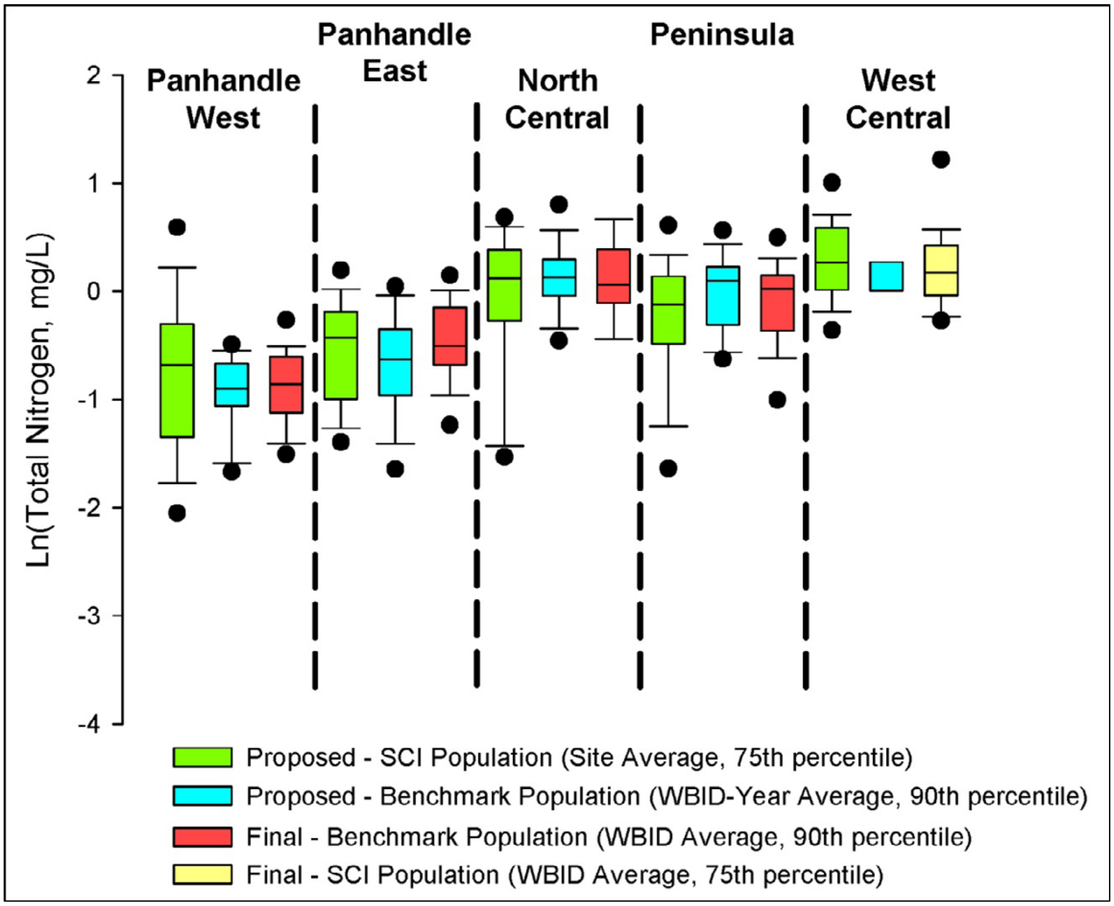

Figure 1-25. Box-whisker plots of TP distributions [Lnn(TP, mg/L)] across EPA's NWRs. Note that the median is indicated by the solid horizontal line within each box; the 25th and 75th percentiles are indicated by the lower and upper horizontal lines of the box, respectively; the 10th and 90th percentiles are indicated by the lower and upper whiskers, respectively; and the 5th and 95th percentiles are indicated by the lower and upper filled circles, respectively. EPA proposed criteria based on the SCI Population using site averages at the 75th percentile of the distribution (light green) and the Benchmark Population using WBID-years at the 90th percentile of the distribution or 75th percentile for the West Central NWR (light blue). EPA's final criteria are based on the Benchmark Population using WBID averages at the 90th percentile of the distribution for the Panhandle West, Panhandle East, North Central, and Peninsula NWRs (light red) and the SCI Population using WBID averages at the 75th percentile of the distribution for the West Central NWR (light yellow).

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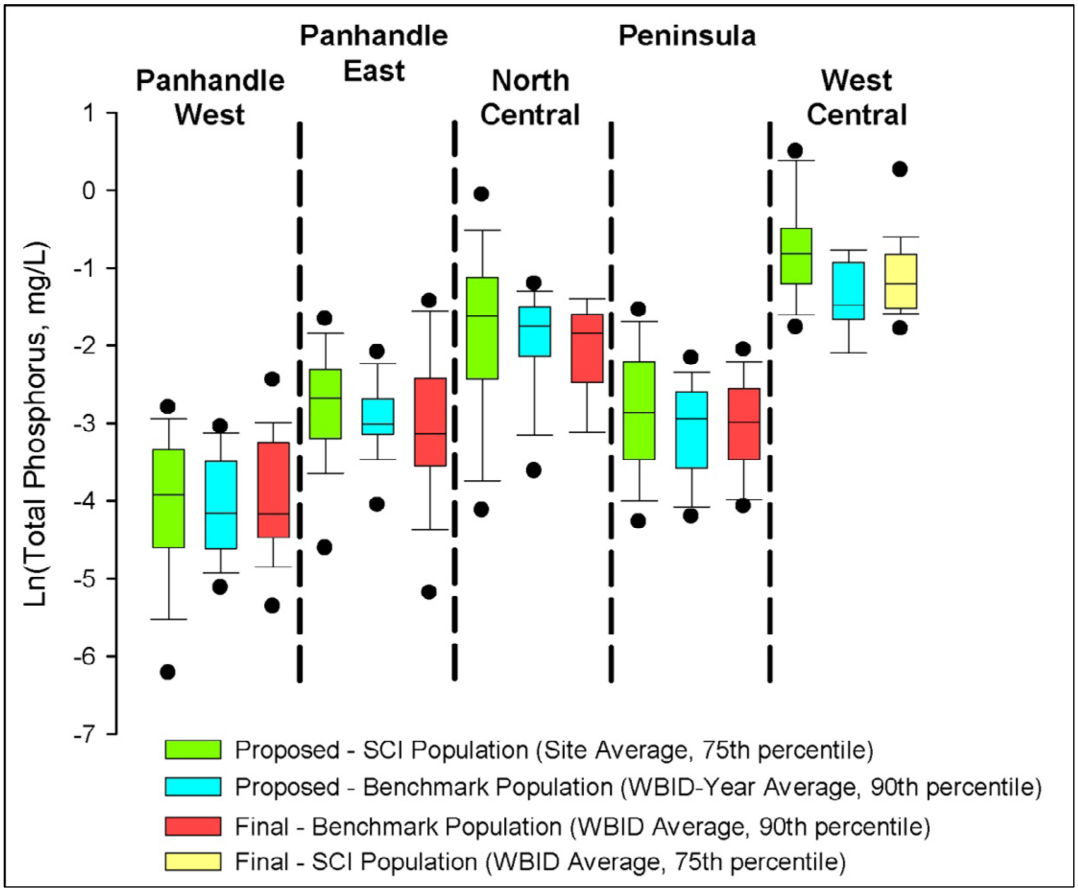

Figure 1-26. Box-whisker plots of TN distributions [Ln(TN, mg/L)] across EPA's NWRs. Note that the median is indicated by the solid horizontal line within each box; the 25th and 75th percentiles are indicated by the lower and upper horizontal lines of the box, respectively; the 10th and 90th percentiles are indicated by the lower and upper whiskers, respectively; and the 5th and 95th percentiles are indicated by the lower and upper filled circles, respectively. EPA proposed criteria based on the SCI Population using site averages at the 75th percentile of the distribution (light green) and the Benchmark Population using WBID-years at the 90th percentile of the distribution or 75th percentile for the West Central NWR (light blue). EPA's final criteria are based on the Benchmark Population using WBID averages at the 90th percentile of the distribution for the Panhandle West, Panhandle East, North Central, and Peninsula NWRs (light red) and the SCI Population using WBID averages at the 75th percentile of the distribution for the West Central NWR (light yellow).

1.4 Defining Healthy Aquatic Life in Florida Streams

The ability to determine the biological condition of a waterbody's aquatic community is critical to informing decisions related to implementation of state and federal water quality programs. Biological assessment measures and tools are very valuable for determining whether, as necessary for this rulemaking, a particular waterbody has a *balanced natural population of aquatic flora and fauna*. They are also important for defining what this means in terms of the expected diversity and abundance of aquatic life and structure and function of the aquatic community.

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This section describes the factors EPA considered in interpreting Florida's definition of a healthy, well-balanced population of natural flora and fauna for purposes of deriving numeric nutrient criteria in Florida's streams. These biological factors were used, in conjunction with other data, to determine the particular nutrient concentrations that might interfere with designated use protection and attainment.

The response of biological communities to anthropogenic point source pollution initially received attention in Florida during the late 1950s. In 1958 Bill Beck, a biologist with the Florida State Board of Health, wrote a series of "*Biological Letters*" in which he introduced the concept of using invertebrates as biological indicators—especially for demonstrating the effects of excess organic matter on streams and lakes (the saprobity index concept). What became known as "*Beck's Biotic Index*" was developed by sampling invertebrates at control sites located upstream of point source discharges and observing which sensitive taxa were eliminated at sites downstream of the effluent sources (Beck 1954).

In the early 1970s and 1980s, benthic invertebrates were routinely sampled with multi-plate artificial substrate samplers (Hester-Dendy samplers). Hester-Dendy samplers are placed in receiving waters for 28 days and data were summarized using the Shannon-Weaver diversity index, a biological metric derived from information theory that became a popular index to communicate complicated biological results. The index is based on a combination of taxa richness at a site and evenness of the distribution of individual abundances. Low diversity scores represent conditions where a few pollution-tolerant organisms are very abundant, to the exclusion of other less pollution tolerant taxa. The index is specified in Florida's water quality standards (WQS) as a measure of biological integrity (Rule 62-302.530, F.A.C.). It has been applied by comparing site-specific control sites to nearby test sites.

Currently, biological data are assessed by comparing them to regional expectations for biological communities, which in turn, are based on samples from reference sites. The latter are selected using regional professional judgment on least-disturbed or minimally disturbed stream locations. Metrics that summarize different characteristics of a biological assemblage are calculated from reference site data. A distribution of reference site metric values is then calculated, and scores are selected to represent expectations for each metric based on the reference site population. Several different metrics are then combined into a dimensionless multi-metric index by assigning points to individual metrics based on their relative similarity to the reference condition and summing across metrics.

To use a biological assessment tool, an understanding of an ecosystem's biological components and sources of variability is important. Aquatic organisms respond to a wide variety of factors, natural and anthropogenic. As organisms integrate and respond to these factors over time, a characteristic community structure emerges, with a range of natural variability. A portion of this natural variability can be explained by random natural events such as floods and drought, which determine the relative abundance of inundated substrates available for invertebrate colonization. These natural stressors (e.g., flood, drought, natural low substrate diversity, periodic natural low dissolved oxygen) can affect all sites—even those with minimal disturbance from humans, to a certain degree.

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1.4.1 Development of the Stream Condition Index

Florida's Stream Condition Index (SCI) was developed in 2004 and revised in 2007. The SCI is a multi-metric index that assesses stream health using the benthic macroinvertebrate community. FDEP expends great efforts to ensure that data are produced with the highest quality, both in the field and in the lab. Samplers and lab technicians follow detailed standard operating procedures (SOPs), and additional guidance for sampling and data use is provided through a FDEP document titled *Sampling and Use of the Stream Condition Index (SCI) for Assessing Flowing Waters: A Primer* (DEP-SAS-001/09). Samplers are approved to conduct the SCI only after passing a rigorous audit with the FDEP, and laboratory taxonomists are regularly tested and must maintain >95% species identification accuracy.

The SCI is composed of 10 metrics, eight of which decrease in response to human disturbance, while the remaining 2 (% very tolerant and % dominant) increase in response to human disturbance. Based on reference site community similarity, three stream bioregions, in which there are slightly different expectations for the metrics based on natural differences, were established: the Panhandle, the Northeast, and the Peninsula (note that the SCI is not calibrated for ecoregion 76, the Southern Florida Coastal Plain, where few natural streams exist) (Griffith et al. 1994). These bioregions are different from NWRs because they reflect natural differences in macroinvertebrate community structure rather than expected nutrient dynamics. For more information on the development of the SCI, see *Appendix A6. FDEP's Development and Testing of Biomonitoring Tools for Macroinvertebrates in Florida Streams*.

1.4.2 Establishing Healthy Biological Conditions in Streams—Stream Condition Index: Application of the Reference Condition Approach

FDEP examined the lower distribution of reference site multi-metric scores to explore such an approach for the potential to establish aquatic life use support thresholds, in combination with the Biological Condition Gradient (BCG) approach (see below), and EPA used the results of this analysis for identifying the threshold between healthy and unhealthy biological conditions.

FDEP conducted statistical interval and equivalence tests with SCI data from 55 streams considered to be reference condition streams, including a portion of verified nutrient benchmark sites with additional data from the Fore et al. (2007) analysis. These tests were used to determine a value for SCI that distinguishes sites that are similar to conditions observed at reference sites from those that are different. The interval and equivalence tests provide probabilistic tests of two different questions: (1) is the site in question within the range of conditions observed at reference sites?, and (2) is the site in question outside of the range of conditions observed at reference sites? Because of uncertainty associated with defining precisely the range of conditions observed at reference sites from a limited number of samples, the answers to these two questions yield two different values of SCI (Table 1-6). The FDEP examination of the two most recent visits at 55 reference streams showed that the 2.5th percentile of reference data was in the range of 35–44 points. The middle of this range was 40 points, which represents an impairment threshold that balanced errors associated with defining whether a site was similar or different from the reference distribution.

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Table 1-6. Results of interval and equivalence tests conducted on reference sites with two SCI results.

Threshold (Description)	Reference Site Mean	Threshold (Numeric)	Different from Reference	Undetermined	Similar to Reference
2.5th percentile of reference	65	40	<35	35–44	>44

Note: shown are site mean, threshold, and range for threshold values defined at the 2.5th percentile of reference sites ($p < 0.05$; $n = 55$ reference sites with two SCI values for each site). Reference site values are taken from Fore et al. (2007) and comprehensively verified nutrient benchmark sites.

When calibrating a threshold between acceptable and unacceptable biological conditions using an index, the amount of human disturbance inherent among reference sites is an important issue. A rigorous reference site selection process provides increased confidence that the reference site population is least-disturbed, or minimally impacted by human influence. No method completely removes uncertainty in setting appropriately protective thresholds. *Appendix A7. FDEP's Site Information and Taxa Lists for SCI BCG Workshop Samples* contains complete taxa data for the samples used in this analysis. *Appendix A8. Memo to FDEP Regarding Interval and Equivalence Tests for the SCI* contains additional information on the interval and equivalence tests.

1.4.3 Biological Condition Gradient Approach

The biological condition gradient (BCG) is a conceptual model that assigns the relative health of aquatic communities into one of six levels (categories), from natural to severely altered (Davies and Jackson 2006, Figure 1-27). The BCG uses biological attributes of aquatic systems that predictably respond to increasing pollution and human disturbance. Although these attributes are measurable, some are not routinely quantified in monitoring programs (e.g., rate measurements such as productivity), but may be inferred through community composition data (e.g., abundance of taxa indicative of organic enrichment). The following biological attributes are considered in the BCG:

- historically documented, sensitive, long-lived or regionally endemic taxa;
- sensitive and rare taxa;
- sensitive but ubiquitous taxa;
- taxa of intermediate tolerance;
- tolerant taxa;
- nonnative taxa;
- organism condition;
- ecosystem functions;
- spatial and temporal extent of detrimental effects; and
- ecosystem connectance.

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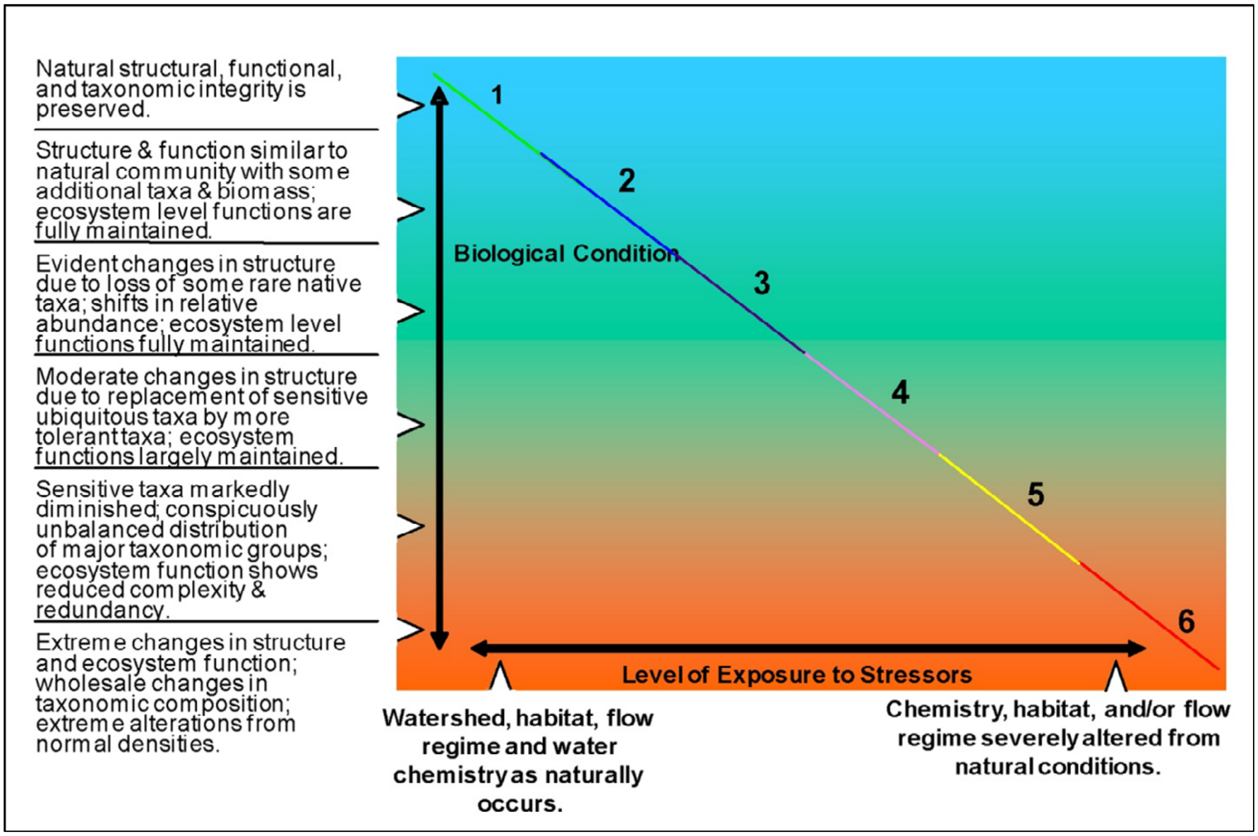

Source: Davies and Jackson (2006).

Figure 1-27. The Biological Condition Gradient (BCG) conceptual model.

As noted above, the gradient represented by the BCG is divided into six levels (tiers) of condition that were defined by consensus (Davies and Jackson 2006) using experienced aquatic biologists from across the United States, including Florida. The six tiers are as follows:

1. Native structural, functional, and taxonomic integrity is preserved; ecosystem function is preserved within range of natural variability.
2. Virtually all native taxa are maintained with some changes in biomass and/or abundance; ecosystem functions are fully maintained within range of natural variability.
3. Some changes in structure due to loss of some rare native taxa; shifts in relative abundance of taxa but sensitive-ubiquitous taxa are common and abundant; ecosystem functions are fully maintained through redundant attributes of the system.
4. Moderate changes in structure due to replacement of some sensitive-ubiquitous taxa by more pollution tolerant taxa, but reproducing populations of some sensitive taxa are maintained; overall balanced distribution of all expected major groups; ecosystem functions largely maintained through redundant attributes.
5. Sensitive taxa are markedly diminished; conspicuously unbalanced distribution of major groups from that expected; organism condition shows signs of physiological stress; system function shows reduced complexity and redundancy; increased buildup or export of unused materials.

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6. Extreme changes in structure; wholesale changes in taxonomic composition; extreme alterations from normal densities and distributions; organism conditioning is often poor; ecosystem functions are severely altered.

The six levels described above are used to correlate biological index scores with biological condition, as part of calibrating an index. FDEP conducted a BCG exercise to calibrate scores for the SCI. Twenty-two experts examined taxa lists from 30 stream sites throughout Florida, 10 in each ecoregion (Fore et al. 2007) that spanned the range of SCI scores (see *Appendix A7. FDEP's Site Information and Taxa Lists for SCI BCG Workshop Samples*). Without knowledge of SCI scores, the experts reviewed the macroinvertebrate assemblage data from each site sample and assigned a BCG score from 1 to 6, where 1 represents natural or native condition and 6 represents a condition severely altered in structure and function from a natural condition. Each expert independently assigned a BCG score to each site and then discussed their scores and scoring rationale. Each expert could make an informed change to their scores based on input from other participants.

1.4.4 Evaluation of BCG Calibration Information

A proportional odds logistic regression model (Guisan and Harrell 2000) was used to estimate the relationship between SCI scores (a continuous variable) and BCG tiers (a categorical variable) (see *Appendix A9. Mapping Continuous Biological Index Values to BCG Tiers*). This model is based on the cumulative probability of a site being assigned to a given BCG tier (e.g., Tier 3) or to any higher quality tier (Tiers 1 and 2). Thus, five parallel models are fit, modeling the probability of assignment to Tiers 5 to 1, Tiers 4 to 1, Tiers 3 to 1, Tiers 2 to 1, and Tier 1 only. Once these five models are fit, the probability of assignment to any single tier can be extracted from the model results.

In Figure 1-28, the mean predictions of the proportional odds logistic regression models are plotted as solid lines. The lines are color-coded and labeled by different tiers, and each line can be interpreted as the proportion of experts that assigned samples with the indicated SCI value to a particular tier. For example, approximately 90% of experts assigned a sample with the lowest SCI score to Tier 6 (brown line), while the remaining experts assigned the sample to Tier 5 (purple line). In Figure 1-28, solid circles represent the actual expert assignments recorded from the workshop for each SCI value. The size of the circle is proportional to the number of experts that assigned a sample to a particular tier, and the circles are color-coded by tier. Expert assignment of BCG scores varied, but there was a central tendency at any given SCI score.

EPA identified the threshold at an SCI score where there was an approximately equally low probability of assignment to Tier 5 (i.e., unhealthy) and a low probability of assignment to Tier 2 (i.e., reference conditions). The resultant threshold balanced the probability of incorrectly assessing a degraded site as representing healthy biological conditions with the probability of incorrectly assessing a reference site as unhealthy. The resulting score of 42 was similar to the threshold of 40 determined by FDEP using the reference site approach.

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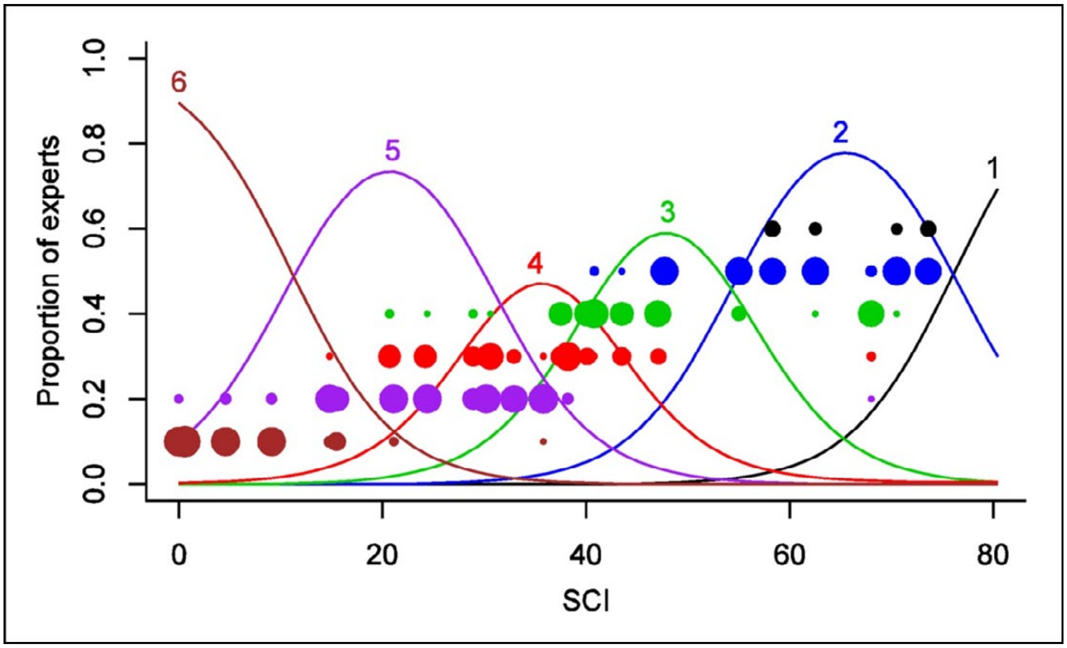

Figure 1-28. Biological Condition Gradient tier assignments modeled with a proportional odds logistic regression.

1.4.5 Selecting the Threshold between Healthy and Unhealthy Biological Conditions

Weighing these multiple lines, EPA selected an SCI value of 40 as the threshold between healthy and unhealthy biological condition. This threshold is supported by the distribution of reference site scores and corresponds with a BCG category midway between Tiers 3 and 4 using the BCG tool. The proportional odds analysis provides assurance that stream communities deemed exceptional (EPA's BCG Tier 2) would not be considered unhealthy at a threshold of 40. Even with these two lines of evidence, uncertainty is associated with a single threshold that defines sites as healthy or unhealthy.

1.4.6 Analysis of Biological Data in the Benchmark Population

As described previously, the SCI is negatively correlated with LDI, and so, biological health decreases in response to increasing levels of human activities (Figure 1-29). However, within the low range of LDI (≤ 2) associated with the nutrient benchmark site data set, no correlation was found between LDI score and SCI score (Figure 1-30). (Note that SCI scores were averaged for sites with more than one SCI sampling event).

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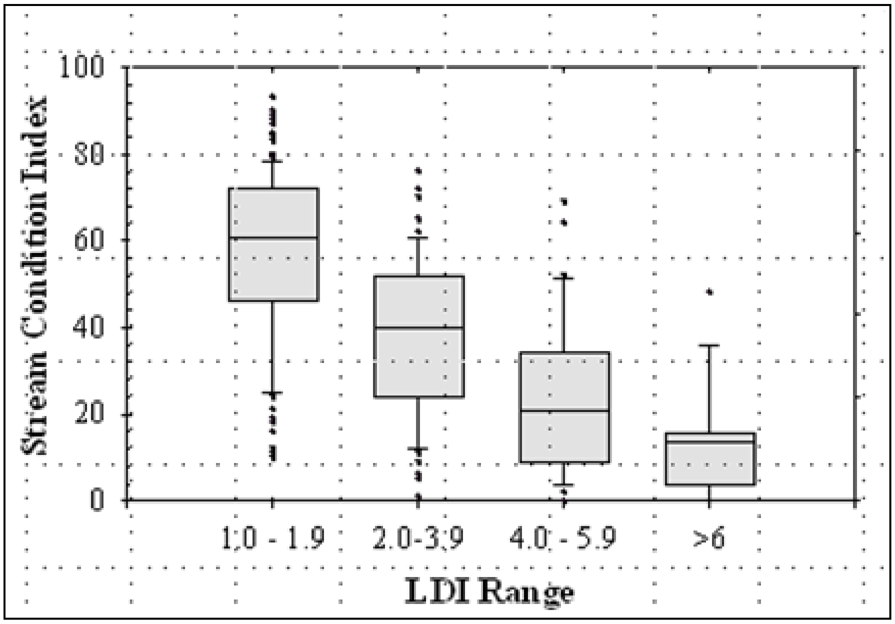
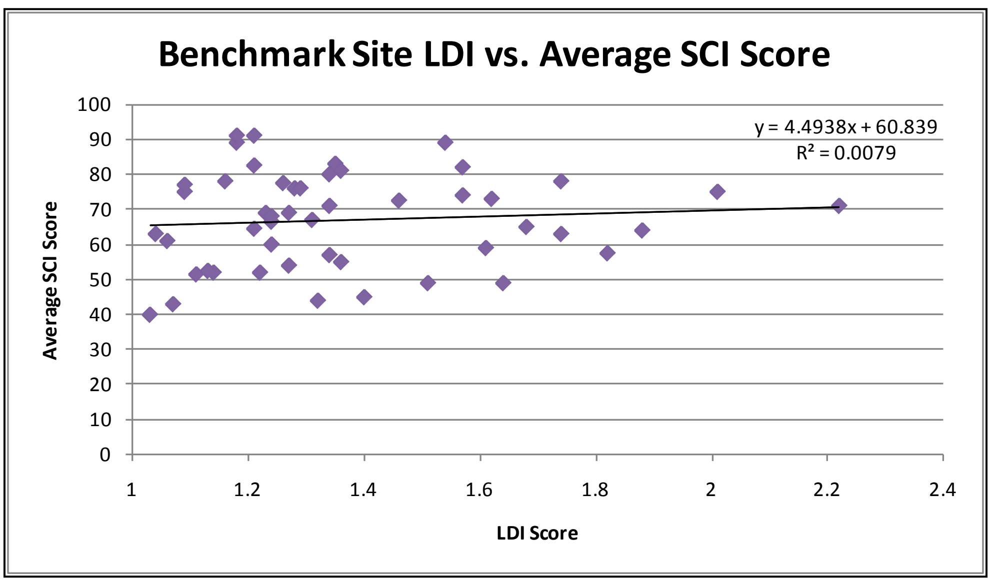


Source: FDEP (2009).

Figure 1-29. Distribution of SCI across the range of corridor LDI scores.



Source: FDEP (2009).

Figure 1-30. Average SCI scores Benchmark Population plotted against corridor LDI scores.

1.5 Duration and Frequency for Streams Numeric Nutrient Criteria

Ambient water quality criteria contain the following three components: magnitude, duration, and frequency. The criterion-duration is expressed as an annual geometric mean. EPA is finalizing the criterion-frequency as a no-more-than-one-in-three-years excursion frequency for the annual geometric mean criteria for streams. The use of the annual geometric mean as the duration

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component of the criteria is consistent with the data set used to derive these criteria, which applied distributional statistics to measures of annual geometric mean values from multiple years of record. As for frequency, EPA has determined that a no-more-than-one-in-three frequency of excursion is consistent with the time frame associated with stream ecosystem recovery from disturbance and, therefore, will not result in unacceptable effects on aquatic life (Chung et al. 1993; Huchens et al. 1998; Minshall 2003; Stephan et al. 1985; Tikkanen et al. 1994; Vieira et al. 2004; Wallace 1990; Wallace et al. 1986, 1991).

Appropriate duration and frequency components of criteria should be based on how the data used to derive the criteria were analyzed and the implications for protecting designated uses given the effects of exposure at the specified criterion concentration for different periods and recurrence patterns.

Lotic systems, such as streams and rivers, flow unidirectionally and, depending on the gradient and other factors, typically have shorter residence times for nutrients than lentic (e.g., ponds, lakes) systems. Lotic systems also have a higher probability that scour may remove algae that is subsequently transported downstream. The likelihood of nutrient effects in these systems is therefore based largely on the prevailing concentration of nutrients as opposed to total loads. Many algae and macrophytes exhibit luxury uptake, in which plants are able to store nutrients in excess of current requirements to support future growth. This complicates the determination of the appropriate averaging period for nutrient concentrations, which may depend on the rates on which algae are able to uptake nutrients.

Frequent disturbance from floods (monthly or more frequently) and associated movement of bed materials can scour algae from the substratum rapidly and often enough to prevent attainment of high biomass (Peterson 1996; Power and Stewart 1987,). In areas with less stable substrata, such as sandy bottomed streams, only slight increases in flow may lead to bed movement and scouring whereas areas with larger, rocks are more resistant to scour and may support higher periphyton biomass for longer periods of time (Cattaneo et al. 1997; Dodds 1991). In either case, where there is frequent movement of substrata, high nutrients may not necessarily translate into excessive algal biomass in the stream reach itself (Biggs et al. 1998a, b).

Low and stable flow conditions should be considered in addition to frequency and timing of floods when considering nutrient criteria for streams. Flood frequency and scouring may be greater in steep gradient (steep slope) and/or channelized streams, and in watersheds subject to intense precipitation. Periods of drying can also reduce algal biomass to low levels (Dodds et al. 1996). A stream may flood frequently during certain seasons, but also remain stable for several months at a time. The effects of eutrophication may be most pronounced during stable low flows. These stable flow periods are generally associated with low flow conditions, which can result in the highest nutrient concentration from point sources discharging relatively constant nutrient loads. Hence, low-flow periods are often the most critical conditions during which algal biomass causes unacceptable effects on aquatic life.

Given that the data compiled and analyzed for developing streams criteria represent all seasons and conditions during the year, and the technical reasons stated above that effects can manifest themselves with a growing season and can vary depending on hydrologic condition, it follows

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that an annual averaging period is the appropriate criterion-duration for the proposed streams criteria.

Nutrient criteria are typically established within a range of natural variability. A temporary spike in nutrient levels does not necessarily harm the aquatic resource. In fact, natural systems have evolved to process variable inputs of nutrients, particularly where there is high natural variability in hydrologic conditions and precipitation patterns (e.g., wet years, dry years). Although biological response in the form of algal production, measured by chl-*a*, can appear very quickly, longer term shifts in biological conditions, such as loss of sensitive species, do not occur as the result of a single event or conditions in a single year. For example, if periphyton experience increased nutrients (due to high runoff) during a portion of a year, the effect on them is not expected to be as large or persistent as if increased nutrient concentrations occur consistently over multiple years. In other words, severe nutrient impacts often result from chronic exposure to elevated nutrients. In addition, nutrient enrichment is more likely to be a chronic stress given the nature of the sources (e.g., lack of appropriate point-source discharge treatment, seasonally land applied fertilizers, regular stormwater runoff, etc.).

The data set used to derive these criteria is based on distributional statistics of geometric mean values from multiple years of record. Data from a central tendency of multiple years of record lend themselves well to expression of an annual average to protect from the chronic effects described above. This supports a criterion-frequency expression of the annual geometric mean. A no more than one in three year frequency of excursions avoids unacceptable effects on aquatic life as it will allow the stream ecosystem enough time to recover from the occasionally elevated year of nutrient loadings (Chung et al. 1993; Huchens et al. 1998; Minshall 2003; Stephan et al. 1985; Tikkanen et al. 1994; Vieira et al. 2004; Wallace 1990; Wallace et al. 1986, 1991).

Frequency and duration components that take into account that hydrological variability will in turn produce variability in measured nutrient concentrations, and individual measurements may exceed the criteria. Individual measurements may exceed the criteria, but isolated nutrient exceedances are generally not evidence of unacceptable effects on aquatic life. Furthermore, these components balance the representation of underlying data and analyses on the basis of central tendency of many years of data (i.e., the annual geometric means component) with the need to exercise some caution to ensure that streams have sufficient time to process individual years of elevated nutrient levels and avoid the possibility of cumulative and chronic effects (i.e., the no more than one in three-year component).

1.6 Summary

EPA developed numeric nutrient criteria for Florida using methods consistent with national nutrient criteria guidance (U.S. EPA 2000) that is general in its methodologies and intended to be adapted to each state's unique data quality and availability issues; unique geologic, climatic, and biological conditions; the nature of biological responses to nutrients, and other factors. Thus, the process described for Florida may not necessarily transfer exactly to any other state and would require a consideration of the above factors and any other information deemed necessary to derive defensible criteria. The general process detailed in the 2000 EPA guidance involves classification, data gathering and preparation, data analysis, and criteria derivation.

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Florida streams were first classified to account for and reduce variability associated with natural gradients in nutrient concentrations due to non-anthropogenic factors (e.g., hydrology, geology). EPA identified five NWRs that were selected based on geological and hydrological considerations, and were shown to explain the most variance in TP concentrations of least-disturbed sites of other candidate classifications. The five NWRs are the Panhandle West, Panhandle East, North Central, Peninsula, and West Central. These NWRs were also used for TN because the statistical modeling indicated that a five region model explains an equivalent amount of variability compared to an alternative regional model for two TN regions. For consistency in application of criteria and in deference to the co-limiting nature of TN and TP, EPA decided that similar regionalization for TN and TP is both defensible and protective of designated uses.

All nutrient concentration data available within the State of Florida in the STORET and GWIS databases were considered and only those data that met data quality standards as required by FDEP QA rule 62-160 and FDEP's "Process for Determining Data Usability" were used (FDEP 2008, see *Appendix A2. FDEP's Process for Assessing Data Usability*). Similarly, biological data that had been collected and quality assured by FDEP were gathered together. These data included data primarily on macroinvertebrates and was used to calculate an FDEP measure of biological condition, the Stream Condition Index (SCI). Land cover data were gathered for the entire state. EPA also considered the EPA-approved list of CWA section 303(d) impaired waters.

EPA derived numeric nutrient criteria using a reference-based distribution approach for TN and TP, but not for chl-*a*. EPA considered deriving chl-*a* criteria for Florida streams using a distribution-based approach, however, the Agency could not identify a specific chl-*a* threshold that would be associated with attaining and maintaining Florida's designated uses for Class I and III streams, and consistent with Florida's current narrative criteria for nutrients. EPA notes that FDEP currently employs a chl-*a* threshold of impairment (0.020 mg/L) for streams in its water quality standards (Impaired Water Rule, Rule 62-303, F.A.C.). Chl-*a* values above this threshold demonstrate that there is an "imbalance" in flora and fauna such that Florida's narrative nutrient criterion is not attained. Florida has not identified an "attainment threshold," or that level of chl-*a* below which a stream is considered in attainment of the narrative criterion.

For the reference based approach, EPA estimated distributional statistics for two principal reference populations—a Benchmark Population represented by sites evaluated as least-disturbed by humans and an SCI Population represented by sites with demonstrated biologically healthy conditions.

For the benchmark approach, reference sites were identified that met the following criteria:

1. LDI score <2 for land use within the 100m corridor 10km upstream of the sample site;
2. Not in WBIDs listed on the EPA-approved Florida CWA section 303(d) impaired waters list for nutrients and/or dissolved oxygen;
3. Average nitrate/nitrite concentrations < 0.35 mg/L;
4. No land uses or nutrient sources adjudged using aerial photographs and FDEP district biologist input that would remove them from consideration as least-impacted sites for nutrients;

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5. Not within WBIDs with average SCI scores <40, and;
6. Watershed or near-field LDI scores <3.

For the SCI Population, reference sites were identified that met the following criteria:

1. Not within WBIDs with average SCI scores <40, and;
2. Not in WBIDs listed on the EPA-approved Florida CWA section 303(d) impaired waters list for nutrients and/or dissolved oxygen.

WBID-averages were calculated for purposes of estimating the frequency distribution of TN and TP by each NWR for the Benchmark and SCI Populations. EPA derived numeric nutrient criteria using a reference-based distribution approach for TN and TP, but not for chl-*a* (Table 1-7). Final nutrient criteria were derived from the 90th percentile of TN and TP concentrations in the Benchmark Population for Panhandle West, Panhandle East, North Central, and Peninsula Regions. The 75th percentile of the SCI population was used for the West Central Region.

Table 1-7. EPA's final numeric nutrient criteria for Florida streams.

Nutrient Watershed Region	Instream Protection Value Criteria	
	TN (mg/L) *	TP (mg/L) *
Panhandle West [a]	0.67	0.06
Panhandle East [b]	1.03	0.18
North Central [c]	1.87	0.30
West Central [d]	1.65	0.49
Peninsula [e]	1.54	0.12

Notes: Watersheds pertaining to each Nutrient Watershed Region (NWR) were based principally on the NOAA coastal, estuarine, and fluvial drainage areas with modifications to the NOAA drainage areas in the West Central and Peninsula Regions that account for unique watershed geologies. For more detailed information on regionalization and which WBIDs pertain to each NWR, see the Technical Support Document.

[a] Panhandle West Region includes: Perdido Bay Watershed, Pensacola Bay Watershed, Choctawhatchee Bay Watershed, St. Andrew Bay Watershed, Apalachicola Bay Watershed.

[b] Panhandle East Region includes: Apalachee Bay Watershed, and Econfina/Steinhatchee Coastal Drainage Area.

[c] North Central Region includes the Suwannee River Watershed.

[d] West Central Region includes: Peace, Myakka, Hillsborough, Alafia, Manatee, Little Manatee River Watersheds, and small, direct Tampa Bay tributary watersheds south of the Hillsborough River Watershed.

[e] Peninsula Region includes: Waccasassa Coastal Drainage Area, Withlacoochee Coastal Drainage Area, Crystal/Pithlachascotee Coastal Drainage Area, small, direct Tampa Bay tributary watersheds west of the Hillsborough River Watershed, Sarasota Bay Watershed, small, direct Charlotte Harbor tributary watersheds south of the Peace River Watershed, Caloosahatchee River Watershed, Estero Bay Watershed, Kissimmee River/Lake Okeechobee Drainage Area, Loxahatchee/St. Lucie Watershed, Indian River Watershed, Daytona/St. Augustine Coastal Drainage Area, St. John's River Watershed, Nassau Coastal Drainage Area, and St. Mary's River Watershed.

* For a given waterbody, the annual geometric mean of TN or TP concentrations shall not exceed the applicable criterion concentration more than once in a three-year period.

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1.7 References

Babbar-Sebens, M., and R. Karthikeyan. 2009. Consideration of sample size for estimating contaminant load reductions using load duration curves. *Journal of Hydrology* 372:118–123.

Baker, L.A., P.L. Brezonik, and C. Kratzer. 1985. Nutrient Loading Models for Florida Lakes. In Lake and Reservoir Management: Practical Applications, eds. J. Taggart and L. Moore, Proc. 4th Annual Meeting. North American Lake Management Society, McAfee, NJ. pp. 253–258.

Barbour, M., J. Gerritsen, and J. White. 1996a. *Development of the Stream Condition Index (SCI) for Florida*. Prepared for Florida Department of Environmental Protection, Tallahassee, Florida by Tetra Tech, Inc., Owings Mills, MD.

Beck, W.M. 1954. Studies in stream pollution biology. I. a simplified ecological classification of organisms. *Quarterly Journal of the Florida Academy of Sciences* 17:212–227.

Biggs, B.J.F., C. Kilroy, and R.L. Lowe. 1998a. Periphyton development in three valley segments of a New Zealand grassland river: Test of a habitat matrix conceptual model within a catchment. *Archiv fuer Hydrobiologie* 143:147–177.

Biggs, B.J.F., R.J. Stevenson, and R.L. Lowe. 1998b. A habitat matrix conceptual model for stream periphyton. *Archiv fuer Hydrobiologie* 143:21–56.

Biggs, B.J.F. 2000. Eutrophication of streams and rivers: dissolved nutrient–chlorophyll relationships for benthic algae. *Journal of the North American Benthological Society* 19:17–31.

Bothwell, M.L. 1985. Phosphorus limitation of lotic periphyton growth rates: an intersite comparison using continuous-flow troughs (Thompson River system, British Columbia). *Limnology and Oceanography* 30:527–542.

Bourassa, N., and A. Cattaneo. 1998. Control of periphyton biomass in Laurentian streams (Quebec). *Journal of the North American Benthological Society* 17:420–429.

Bowling, L.C., and P.D. Baker. 1996. Major cyanobacterial bloom in the Barwon-Darling River, Australia, in 1991, and underlying limnological conditions. *Marine and Freshwater Research* 47:643–657.

Bricker, S.B., C.G. Clement, D.E. Pirhalla, S.P. Orlando, and D.R.G. Farrow. 1999. *National Estuarine Eutrophication Assessment: Effects of Nutrient Enrichment in the Nation's Estuaries*. NOAA, National Ocean Service, Special Projects Office and the National Centers for Coastal Ocean Science, Silver Spring, MD.

Bricker, S., J.G. Ferreira, and T. Simas. 2003. An integrated methodology for assessment of estuarine trophic status. *Ecological Modeling* 169:39–60.

Bricker, S., B. Longstaff, W. Dennison, A. Jones, K. Boicourt, C. Wicks, and J. Woerner. 2007. *Effects of Nutrient Enrichment in the Nation's Estuaries: A Decade of Change*. NOAA Coastal Ocean Program Decision Analysis Series No. 26. National Oceanographic and Atmospheric Administration, National Centers for Coastal Ocean Science, Silver Spring, MD.

Brown, M.T., and K.C. Reiss. 2006. *Proposed Breakpoint of LDI < 2.0 for Determining Minimally Affected Reference Conditions for Water Bodies*. Technical report submitted to the Florida Department of Environmental Protection, and University of Florida, Center for Environmental Policy, Department of Environmental Engineering Sciences, Gainesville, FL.

Brown, M.T., and M.B. Vivas. 2003. *A Landscape Development Intensity Index*. Technical report submitted to the Florida Department of Environmental Protection. University of Florida, Center for Environmental Policy, Department of Environmental Engineering Sciences, Gainesville, FL.

Brown, M.T., and M.B. Vivas. 2005. Landscape development intensity Index. *Environmental Monitoring and Assessment* 101:289–309.

Canfield, D.E., Jr. and R.W. Bachmann. 1981. Prediction of total phosphorus concentrations, chlorophyll *a*, and Secchi depths in natural and artificial lakes. *Canadian Journal of Fisheries and Aquatic Science* 38:414–423.

Cathcart, J.B. 1985. Economic Geology of the Land-Pebble Phosphate District of Florida and its Southern Extension. In *Florida Land-Pebble Phosphate District: Geological Society of America Annual Meeting, field trip guidebook*, eds. Cathcart, J.B., and Scott, T.M., Orlando, FL. pp. 4–27.

Cattaneo, A., T. Kerimian, M. Roberge, and J. Marty. 1997. Periphyton distribution and abundance on substrata of different size along a gradient of stream trophy. *Hydrobiologia* 354:101–110.

Chung, K., J.B. Wallace, and J.W. Grubaugh. 1993. The impact of insecticide treatment on abundance, biomass and production of litterbag fauna in a headwater stream: a study of pretreatment, treatment and recovery. *Limnologica* 28(2): 93–106.

Cohn, T.A., L.L. Delong, E.J. Gilroy, R.M. Hirsch, and D.K. Wells. 1989. Estimating constituent loads. *Water Resources Research* 25:937–942.

Conley, D.J., H.W. Paerl, R.W. Howarth, D. F. Boesch, S.P. Seitzinger, K.E. Havens, C. Lancelot, and G.E. Likens. 2009. Controlling Eutrophication: Nitrogen and Phosphorus. *Science* 323:1014–1015.

Cross, W.F., J.B. Wallace, A.D. Rosemond, and S.L. Eggert. 2006. Whole-system nutrient enrichment increases secondary production in a detritus-based ecosystem. *Ecology* 87:1556–1565.

US EPA ARCHIVE DOCUMENT

US EPA ARCHIVE DOCUMENT

Davies, T.T. 1997. Establishing Site Specific Aquatic Life Criteria Equal to Natural Background. Memorandum to Water Division Directors, Regions 1-10, State and Tribal Water Quality Management Program Directors. U.S. EPA, Office of Water, Office of Science and Technology.

Davies, S.P., and S.K. Jackson. 2006. The biological condition gradient: A descriptive model for interpreting change in aquatic systems. *Ecological Applications* 16:1251–1266.

Dodds, W.K. 1991. Factors associated with dominance of the filamentous green alga *Cladophora glomerata*. *Water Research* 25:1325–1332.

Dodds, W.K., and D.A. Gudder. 1992. The ecology of Cladophora. *Journal of Phycology* 28:415–427.

Dodds, W.K., J.M. Blair, G.M. Henebry, J.K. Keolliker, R. Ramundo, and C.M. Tate. 1996. Nitrogen transport from tallgrass prairie watersheds. *Journal of Environmental Quality* 25:973–981.

Dodds, W.K., W.W. Bouska, J.L. Eitzmann, T.J. Pilger, K.L. Pitts, A.J. Riley, J.T. Schloesser, and D.J. Thornbrugh. 2009. Eutrophication of U.S. freshwaters: Analysis of potential economic damages. *Environmental Science and Technology* 43(1):12–19.

Elser, J.J., M.E.S. Bracken, E.E. Cleland, D.S. Gruner, W.S. Harpole, H. Hillebrand, J.T. Ngai, E.W. Seabloom, J.B. Shurin, and J.E. Smith. 2007. Global analysis of nitrogen and phosphorus limitation of primary producers in freshwater, marine and terrestrial ecosystems. *Ecology Letters* 10:1135–1142.

Elwood, J.W., J.D. Newbold, A.F. Trimble, and R.W. Stark. 1981. The limiting role of phosphorus in a woodland stream ecosystem: effects of P enrichment on leaf decomposition and primary producers. *Ecology* 62:146–158.

FDEP (Florida Department of Environmental Protection). 2005. *Water Quality Assessment Report, Apalachicola-Chipola*. Florida Department of Environmental Protection, Division of Water Resource Management, Tallahassee, FL.

FDEP (Florida Department of Environmental Protection). 2008. *Process for Assessing Data Usability*. DEP-EA001/07. Florida Department of Environmental Protection, Tallahassee, FL.

FEDP (Florida Department of Environmental Protection). 2009. *Technical Support Document: Development of Numeric Nutrient Criteria for Florida Lakes and Streams*. Florida Department of Environmental Protection, Tallahassee, FL. June 2009. <http://www.dep.state.fl.us/water/wqssp/nutrients/docs/tsd_nutrient_crit.docx>. Accessed September 6, 2010.

Fernald, E.A., and E.D. Purdum. 1998. *Water Resources Atlas of Florida*. Florida State University, Institute of Science and Public Affairs, Tallahassee, FL.

Fore, L.S. 2004. *Development and Testing of Biomonitoring Tools for Macroinvertebrates in Florida Streams*. Final report to the Florida Department of Environmental Protection, Tallahassee, FL.

Fore, L.S, R.B. Frydenborg, D. Miller, T. Frick, D. Whiting, J. Espy, and L. Wolfe. 2007. *Development and Testing of Biomonitoring Tools for Macroinvertebrates in Florida Streams*. Florida Department of Environmental Protection, Tallahassee, FL.

Francoeur, S.N. 2001. Meta-analysis of lotic nutrient amendment experiments: detecting and quantifying subtle responses. *Journal of the North American Benthological Society* 20:358–368.

Gaiser, E.E., J.C. Trexler, J.H. Richards, D.L. Childers, D. Lee, A.L. Edwards, L.J. Scinto, K. Jayachandran, G.B. Noe, R.D. Jones. 2005. Cascading ecological effects of low-level phosphorus enrichment in the Florida Everglades. *Journal of Environmental Quality* 34:717–723.

Gao, X. 2006. *Nutrient and Unionized Ammonia TMDLs for Lake Jessup, WBIDs 2981 and 2981A*. Florida Department of Environmental Protection, Division of Water Resource Management, Bureau of Watershed Management, Tallahassee, FL.

Greening, H., and A. Janicki. 2006. Toward reversal of eutrophic conditions in a subtropical estuary: water quality and seagrass response to nitrogen loading reduction in Tampa Bay, FL. *Environmental Management* 38:163–178.

Griffith, G.E., J.M. Omernik, C.M. Rohm, and S.M. Pierson. 1994. *Florida Regionalization Project*. U.S. Environmental Protection Agency, Environmental Research Laboratory, Corvallis, OR.

Grubbs, G. 2001. U.S. EPA. (Memorandum to Directors of State Water Programs, Directors of Great Water Body Programs, Directors of Authorized Tribal Water Quality Standards Programs and State and Interstate Water Pollution Control Administrators on Development and Adoption of Nutrient Criteria into Water Quality Standards. November 14, 2001.

Grumbles, B.H. 2007. U.S. EPA. (Memorandum to Directors of State Water Programs, Directors of Great Water Body Programs, Directors of Authorized Tribal Water Quality Standards Programs and State and Interstate Water Pollution Control Administrators on Nutrient Pollution and Numeric Water Quality Standards. May 25, 2007.

Guildford, S.J., and R.E. Hecky. 2000. Total nitrogen, total phosphorus, and nutrient limitation in lakes and oceans: Is there a common relationship? *Limnology and Oceanography* 45:1213–1223.

Guisan, A., and F.E. Harrell. 2000. Ordinal response regression models in ecology. *Journal of Vegetation Science* 11:617–626.

US EPA ARCHIVE DOCUMENT

Hoos, A.B., and G. McMahon. 2009. Spatial analysis of instream nitrogen loads and factors controlling delivery to streams in the southeastern United States using spatially referenced regression on watershed attributes (SPARROW) and regional classification frameworks. *Hydrological Processes* 23.

Hoos, A.B., S. Terziotti, G. McMahon, K. Savvas, K.C. Tighe, and R. Alkons-Wolinsky. 2008. *Data to Support Statistical Modeling of Instream Nutrient Load Based on Watershed Attributes, Southeastern United States*. 2002: U.S. Geological Survey Open-File Report 2008–1163. U.S. Geological Survey, Reston, VA.

Howarth, R.W., A. Sharpley, and D. Walker. 2002. Sources of nutrient pollution to coastal waters in the United States: Implications for achieving coastal water quality goals. *Estuaries* 25(4b):656–676.

Hutchens, J.J., K. Chung, and J.B. Wallace. 1998. Temporal variability of stream macroinvertebrate abundance and biomass following pesticide disturbance. *Journal of the North American Benthological Society* 17:518–534.

Jacobs, T.C., and J.W. Gilliam. 1985. Riparian losses of nitrate from agricultural drainage waters. *Journal of Environmental Quality* 14:472–478.

Jordan, T.E., D.L. Correll, and D.E. Weller. 1993. Nutrient interception by a riparian forest receiving inputs from adjacent cropland. *Journal of Environmental Quality* 22:467–473.

Kilgour, B.W., K.M. Somers, and D.E. Matthews. 1998. Using the normal range as a criterion for ecological significant in environmental monitoring and assessment. *Ecoscience* 5:542–550.

Larsen, D.P., and H.T. Mercier. 1976. Phosphorus retention capacity of lakes. *Journal of the Fisheries Research Board of Canada* 33:1742–1750.

Lowrance, R. 1992. Groundwater nitrate and denitrification in a coastal plain riparian soil. *Journal of Environmental Quality* 21:401–405.

Lowrance, R. 1997. Water quality functions of riparian forest buffers in Chesapeake Bay watersheds. *Environmental Management* 21:687–712.

McMahon, G., L. Tervelt, and W. Donehoo. 2007. *Methods for Estimating Annual Wastewater Nutrient Loads in the Southeastern United States*. U.S. Geological Survey Open-File Report 2007–1040. U.S. Geological Survey, Reston, VA. <http://pubs.usgs.gov/of/2007/1040/NADP>. Accessed August 2007.

Minshall, G.W. 2003. Responses of stream benthic macroinvertebrates to fire. *Forest Ecology and Management* 178:155–161.

Moss, B., I. Hooker, H. Balls, and K. Manson. 1989. Phytoplankton distribution in a temperate floodplain lake and river system. I. hydrology, nutrient sources and phytoplankton biomass. *Journal of Plankton Research* 11:813–835.

US EPA ARCHIVE DOCUMENT

US EPA ARCHIVE DOCUMENT

Mulholland, P.J., and J.R. Webster. 2010. Nutrient dynamics in streams and the role of J-NABS. *Journal of the North American Benthological Society* 29:100–117.

NADP (National Atmospheric Deposition Program). 2008. *National Atmospheric Deposition Program 2007 Annual Summary*. NADP Data Report 2008-01. National Atmospheric Deposition Program, University of Illinois at Urbana-Champaign, Illinois State Water Survey, Champaign, IL.

NRC (National Research Council). 2000. Clean Coastal Waters: Understanding and Reducing the Effects of Nutrient Pollution. Report prepared by the Ocean Study Board and Water Science and Technology Board, Commission on Geosciences, Environment and Resources, National Resource Council. National Academy Press, Washington, DC.

Niu, X. 2004. *Change Point Analysis of the Chlorophyll a and LDI Data from the IWRM Status Network Cycle 1 Lakes Study*. Technical report submitted to the Florida Department of Environmental Protection. Florida State University, Department of Statistics, Tallahassee, FL.

Notestein, S.K., T.K. Frazer, M.V. Hoyer, and D.E. Canfield, Jr. 2003. Nutrient limitation of periphyton in a spring- fed, coastal stream in Florida, USA. *Journal of Aquatic Plant Management* 41:57–60.

Paerl, H.W. 2009. Controlling Eutrophication along the Freshwater–Marine Continuum: Dual Nutrient (N and P) Reductions are Essential. *Estuaries and Coasts* 32:593-601.

Peterjohn, W.T., and D.L. Corell. 1985. Nutrient dynamics in an agricultural watershed: observations on the role of a riparian forest. *Ecology* 65:1466–1475.

Peterson, B.J., J.E. Hobbie, A.E. Hershey, M.A. Lock, T.E. Ford, J.R. Vestal, V.L. McKinley, M.A.J. Hullar, M.C. Miller, R.M. Ventullo, and G.S. Volk. 1985. Transformation of a tundra river from heterotrophy to autotrophy by addition of phosphorus. *Science* 229:1383–1386.

Peterson, C.G. 1996. Mechanisms of lotic microalgal colonization following space-clearing disturbances at different spatial scales. *Oikos* 77:417–435.

Power, M.E., and A.J. Stewart. 1987. Disturbance and recovery of an algal assemblage following flooding in an Oklahoma stream. *American Midland Naturalist* 117:333–345.

Rosemond, A.D., P.J. Mulholland, and J.W. Elwood. 1993. Top-down and bottom-up control of stream periphyton: effects of nutrients and herbivores. *Ecology* 74:1264–1280.

Rosemond, A.D., C.M. Pringle, A. Ramirez, and M.J. Paul. 2001. A test of top-down and bottom-up control in a detritus-based food web. *Ecology* 82:2279–2293.

Rosemond, A D., C.M. Pringle, A. Ramirez, M.J. Paul, and J.L. Meyer. 2002. Landscape variation in phosphorus concentration and effects on detritus-based tropical streams. *Limnology and Oceanography* 47:278–289.

Schindler, D.W. 1974. Eutrophication and recovery in experimental lakes: Implications for lake management. *Science* 184: 897–899.

Schwarz, G.E., A.B. Hoos, R.B. Alexander, and R.A. Smith. 2006. *The SPARROW Surface Water-Quality Model—Theory, Application and User Documentation.* U.S. Geological Survey Techniques and Methods, Book 6. U.S. Geological Survey, Reston, VA.

Shannon, E.E., and P.L. Brezonik. 1972. Relationships between lake trophic state and nitrogen and phosphorus loading rates. *Environmental Science and Technology* 6:719–725.

Slavik, K., B.J. Peterson, L.A. Deegan, W.B. Bowden, A.E. Hershey, and J.E. Hobbie. 2004. Long-term responses of the Kuparuk River ecosystem to phosphorus fertilization. *Ecology* 85:939–954.

Smith, V.H., S.B. Joye, and R.W. Howarth. 2006. Eutrophication of freshwater and marine ecosystems. *Limnology and Oceanography* 51:351–355.

Smith, V.H. 2003. Eutrophication of freshwater and coastal marine ecosystems. *Environmental Science and Pollution Research* 10(2):126–139.

Stephan, C.E., D.I. Mount, D.J. Hansen, J.H. Gentile, G.A. Chapman, and W.E. Brungs. 1985. *Guidelines for Deriving Numeric National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses*. EPA PB85-227049\.

Steward, J.S., and E.F. Lowe. 2010. General empirical models for estimating nutrient load limits for Florida's estuaries and inland waters. *Limnology and Oceanography* 55(1):433–445.

Tikkanen, P., Laasonen, P., Muotka, T., Huhta, A., and Kuusela, K. 1994. Short-term recovery of benthos following disturbance from stream habitat rehabilitation. *Hydrobiologia* 273:121–130.

U.S. EPA (U.S. Environmental Protection Agency). 2000. *Nutrient Criteria Technical Guidance Manual: Rivers and Streams*. EPA-822-B-00-002. U.S. Environmental Protection Agency, Office of Water, Washington, DC.

U.S. EPA (U.S. Environmental Protection Agency). 2007. *Hypoxia in the Northern Gulf of Mexico: An Update by the EPA Science Advisory Board*. EPA-SAB-08-003. EPA Science Advisory Board, Washington, DC.

U.S. EPA (U.S. Environmental Protection Agency). 2010a. *Water Quality Standards for the State of Florida's Lakes and Flowing Waters. Proposed Rule. January 14, 2010*. EPA-HQ-OW-2009-0596, FRL-9105-1.

U.S. EPA (U.S. Environmental Protection Agency). 2010b. *Water Quality Standards for the State of Florida's Lakes and Flowing Waters; Supplemental Notice of Data Availability. August 3, 2010.* EPA-HQ-OW-2009-0596, FRL-9185-2.

US EPA ARCHIVE DOCUMENT

Vieira N.K.M, W.H. Clements, L.S. Guevara, and B.F. Jacobs. 2004. Resistance and resilience of stream insect communities to repeated hydrologic disturbances after a wildfire. *Freshwater Biology* 49:1243–1259.

Wallace, J.B. 1990. Recovery of lotic macroinvertebrate communities from disturbance. *Environmental Management* 14:605–620.

Wallace, J.B., D.S. Vogel, and T.F. Cuffney. 1986. Recovery of a headwater stream from an insecticide induced community disturbance. *Journal of the North American Benthological Society* 5:115–l26.

Wallace, J.B., A.D. Huryn, and G.J. Lugthart. 1991. Colonization of a headwater stream during three years of seasonal insecticidal applications. *Hydrobiologia* 211:65–76.

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Chapter 2: Derivation of EPA's Numeric Nutrient Criteria for Lakes

2.1 Background

Lakes are a highly valued national resource, but especially in Florida, which has a high abundance and unique diversity of lakes. Some lakes are popular for recreational activities such as trophy fishing, swimming, and other watersports; other lakes are sought out for wildlife viewing and wilderness experience.

Florida has an exceptionally large number of lakes (more than 7700 lakes greater than 10 acres). The lakes range in size from very small ponds to Lake Okeechobee, the seventh-largest fresh water lake in surface area in the United States (681 sq. miles). Florida's lakes comprise extreme ranges in water alkalinity, color, and productivity, including crystal-clear ponds and lakes in the pine forests; deeply stained (tea-colored) or blackwater lakes; and hard-water lakes with moderate to high natural productivity. Although most Florida lakes are shallow, there are also a small number of very deep sinkhole lakes. Most of Florida's lakes are natural, formed by solution of limestone into depressions and sinkholes.

Nutrients, especially nitrogen and phosphorus, are essential to life and are not inherently harmful or toxic at natural concentrations. Lakes can be classified as oligotrophic (low in nutrients and productivity), mesotrophic (intermediate in nutrients and productivity), or eutrophic (high in nutrients and productivity). Oligotrophic lakes generally have low productivity and low biomass, yet often have high species diversity. Eutrophic lakes have high productivity and high biomass, but may be characterized by very high abundances of few species. Underlying geology, the character and size of the watershed, and other natural factors can also determine the trophic status of a waterbody.

Anthropogenic activities, such as sewage discharge, soil erosion, agricultural and urban runoff can increase the flow of nutrients into a lake. These activities impact lakes by stimulating excess algal growth, including harmful algal blooms, and reducing water clarity (Carlson 1977; Downing et al. 2001; Elser et al. 1990; Elser et al. 2007; NAS 1969; Paerl 1988; Schindler et al. 1973; Schindler 1974; Smith et al. 1999; Smith et al. 2006; Vollenweider 1968; Vollenweider 1976). This process is termed cultural eutrophication, and can occur over a short time (few years). Natural eutrophication, where a lake's trophic state increases due to natural inflow of sediment, nutrients, and organic matter from its watershed, is a process that may take thousands of years, if it occurs at all (U.S. EPA 2000). Cultural eutrophication is usually reversible, and lakes may recover if the artificial nutrient loads are reduced (e.g., Edmondson 1994).

In Florida, several well-known lakes have been subject to visible and well-documented cultural eutrophication (e.g., Lake Okeechobee, Lake Apopka). Nutrient reductions plans and TMDLs have been developed for these lakes in response to human nutrient enrichment, and some are beginning to respond to the nutrient load reductions (e.g., Coveney et al. 2005). EPA's development of water quality criteria for nitrogen and phosphorus is intended to establish concentrations that are protective; that is, to prevent impairments of the designated use.

EPA recognizes that nutrient limitation of either TN or TP, or co-limitation by TN and TP, is a phenomenon previously observed in many lakes (Schindler 1974). Primary production in fresh water systems can be limited by available nitrogen or phosphorus, or can be co-limited by both nutrients simultaneously. Early eutrophication studies clearly demonstrated phosphorus limitation and eutrophication by phosphorus enrichment in fresh water (e.g., Schindler 1974). Recent studies have shown that nitrogen limitation and co-limitation are more important in fresh water, and phosphorus-limitation is more important in saltwater, than previously recognized (e.g., Guildford and Hecky 2000, Brown et al. 2000, Smith 2006, Elser et al. 2007, Elser et al. 2009). For any given lake, however, it is difficult to assess *a priori* which nutrient is the limiting factor. EPA's approach to deriving numeric nutrient criteria is not premised on the limiting nutrient, but rather a conservative view that both nutrients influence excess algal growth over and that influence may vary over space and time. EPA has expressed its recommended approaches in deriving numeric nutrient criteria for both TN and TP previously for lakes (U.S. EPA 2000) and as a general policy matter (Grubbs 2001; Grumbles 2007). The scientific literature concluding that the source of many adverse water quality impacts can be traced to degradation of water quality upstream of the impacted aquatic system by nutrient pollution (e.g., U.S. EPA 2007, Paerl, H.W. 2009, Conley et al. 2009) further affirms EPA's approach water quality management, through numeric nutrient criteria, of both nitrogen and phosphorus.

This chapter describes the background, methodology, and results of EPA's analysis for deriving the proposed numeric nutrient criteria for Florida lakes. The methodology includes developing lake classification and deriving chl-*a* and nutrient criteria for each of the resultant lake classes. All statistical data analysis was based on Florida's IWR database, as developed and screened by FDEP. An analytical plan describing EPA's numeric nutrient criteria development process for Florida's lakes is provided in Figure 2-1.

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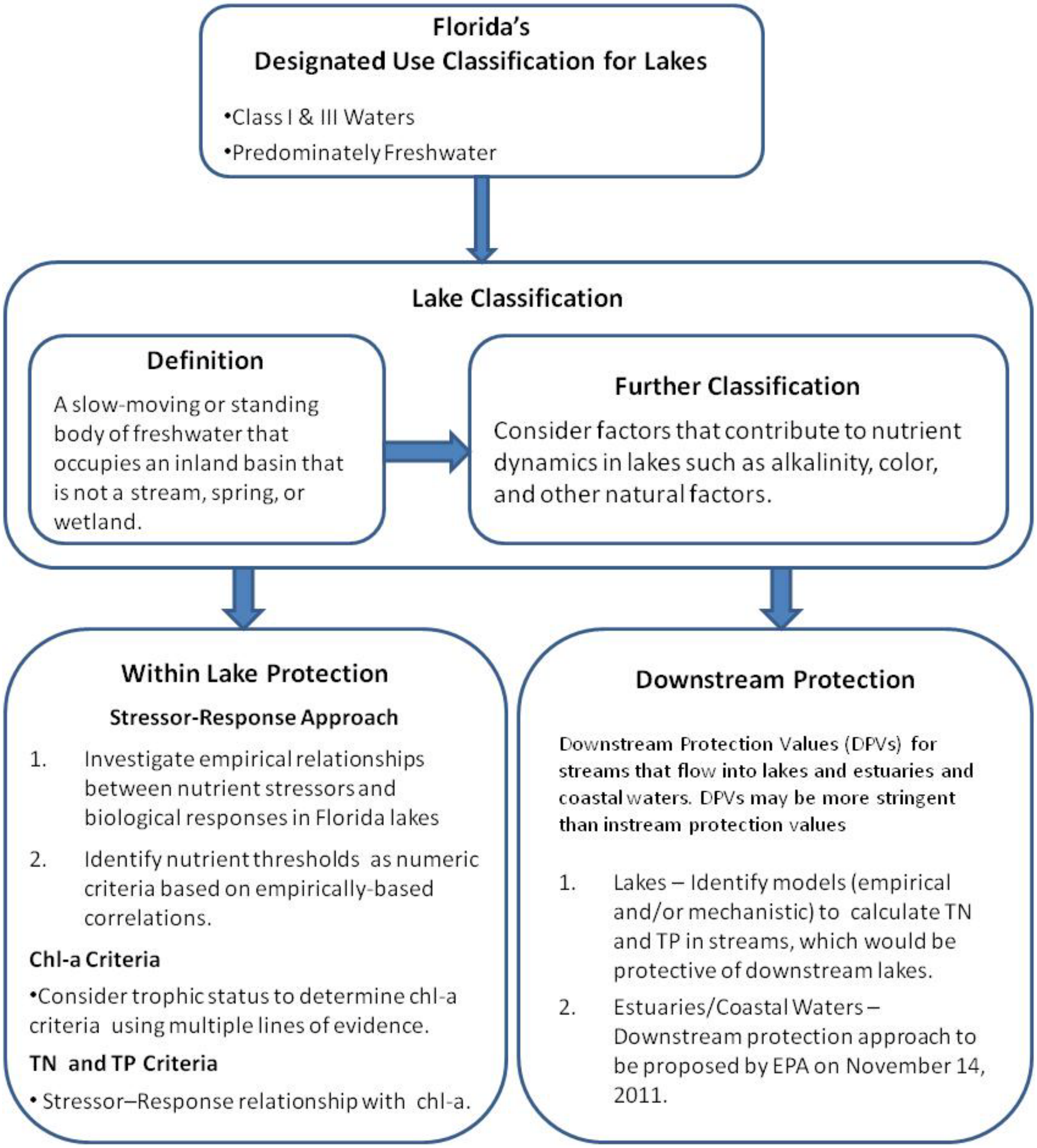

Figure 2-1. Analytical plan of EPA's numeric nutrient criteria development process for Florida's lakes.

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2.2 Lake Classification

2.2.1 Background

Lake classification of lakes may be based on continuous variables such as water color, alkalinity, elevation, latitude; or continuous variables that may be divided into categories such as clear lakes, mountain lakes, geographic regions (U.S. EPA 1998, 2000;). In the case of Florida lakes, EPA sought a classification that predicts inherent, natural differences among lakes in their natural concentrations of chl-*a*, and in the response of planktonic algae to anthropogenic nutrient loading. There are many factors that may influence planktonic algal growth, abundance, and biomass (measured as chl-*a*), including water color, turbidity, alkalinity, N:P ratio, and TN and TP.

Previous studies on Florida lakes identified the following four lake classes: (1) clear, soft water (low alkalinity) lakes; (2) colored, low alkalinity lakes; (3) clear, alkaline lakes; and (4) colored, alkaline lakes. This classification system was originally proposed by Shannon and Brezonik (1972) on the basis of cluster analysis of lakes in north and central Florida. The classification system was subsequently confirmed by Gerritsen et al. (2000) using principal components analysis of a larger statewide data set compiled by Griffith et al. (1997). Griffith and colleagues also developed a geographic classification system of Florida lakes, and identified 47 lake regions using geology, lake origin, and water chemistry. Among the lake groups, clear, softwater (low alkalinity) lakes of northwestern and central sandhills were identified as extremely oligotrophic (Canfield et al. 1983). More recently, Lowe et al. (2009) developed a classification system of northeastern Florida lakes using continuous variables of depth, water color, and alkalinity. The latter two variables are discussed below. These alternative classifications are compared in Section 2.2.4.

To help guide the selection of variables for use in classification, EPA developed a conceptual model linking human activities, stressors (including increased nutrient concentrations), and designated uses (see Figure 2-2). Using this conceptual model, EPA identified geological characteristics (as measured by alkalinity), color, and temperature as factors that may influence estimates of the response of algae to nutrient loading in Florida's lakes.

Increased water color is due to dissolved organic carbon from decaying plant material in forests and wetlands of a lake's surface watershed. Water color in Florida lakes ranges from none (clear) to heavily stained, tea-colored (often called blackwater). Color limits light penetration into the water column, and thus algal growth and biomass (chl-*a*) are limited in heavily colored lakes.

Total alkalinity is a measure of the total concentration of bases in water, which is typically expressed as $CaCO_3$/mg/L. Those bases typically occur as bicarbonate (HCO_3^-) and carbonate ($CO_3^=$), which act as buffers to prevent drastic changes in pH. According to a survey of 946 Florida lakes, the total alkalinity ranges from 0.24 to 552 mg/L, with a median value of 40 mg/L (Lazzarino et al. 2009). The alkalinity of Florida lakes is regulated by the contribution of groundwater that has been in contact with limestone or calcareous soils (Stauffer and Canfield 1992). For example, low alkalinity sandhill lakes often have no inlet or outlet but receive shallow groundwater from siliceous sand soils, and the shallow groundwater is perched above deeper limestone aquifers (Stauffer and Canfield 1992). Other lakes are not isolated from limestone aquifers, and groundwater contributes alkalinity from calcium carbonate dissolved from limestone.

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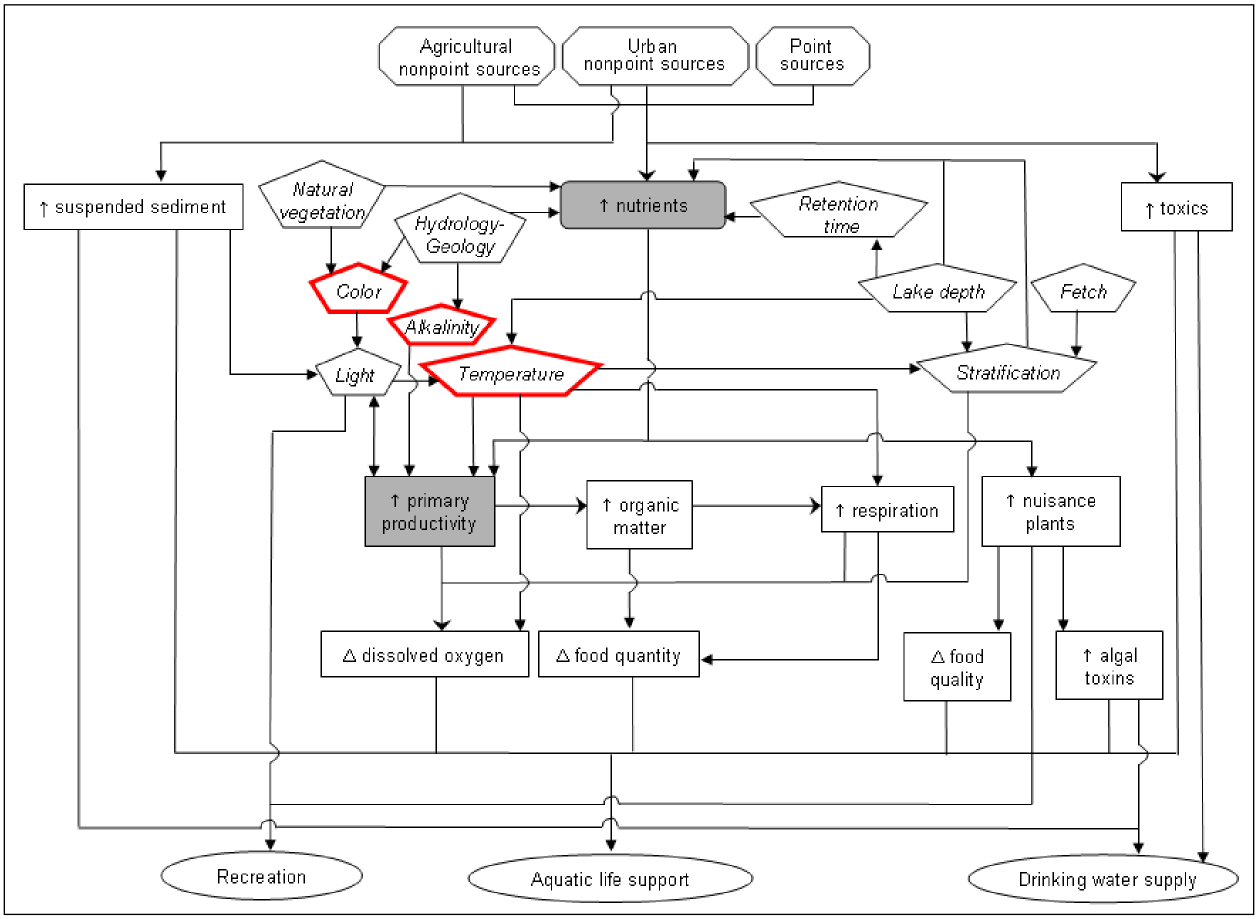

Figure 2-2. Conceptual model showing linkages between nutrient concentrations and designated uses.
Note that boxes filled in gray are the variables for which numeric nutrient criteria are derived: the nutrient variables (TN and TP) and the response variable (primary productivity as quantified by chl-*a*. Shapes outlined in red are alternate pathways linking nutrient concentration to chl-*a* (including these pathways in the classification improves the accuracy of estimate nutrient stressor-response relationships).

EPA's review of the scientific literature found that higher alkalinity lakes are considered to be more productive than lower alkalinity lakes, and alkalinity is associated with productivity of undisturbed reference condition lakes (Lowe et al. 2009; Oglesby 1977; Ryder et al. 1974; Vighi and Chiaudani 1985). Limestone provides carbonate equal to that from atmospheric CO_2 and can also contribute phosphorus to ground and surface waters. The effect of alkalinity has also been shown in experimental studies; increasing alkalinity from 13 to > 50 mg/L $CaCO_3$ increased primary productivity for the same nutrient loading (Arce and Boyd 1975). Alkalinity was also found to influence the species richness and composition of both aquatic macrophytes and algae species (Vestergaard and Sand-Jensen 2000).

Alkalinity has also been used to classify lakes. For example, to comply with the European Water Framework Directive, Danish lakes were classified by depth and alkalinity; the alkalinity threshold used was 10 mg/L $CaCO_3$ (Søndergaard et al. 2005). Similarly, FDEP classified Florida lakes by alkalinity in support of the State's proposed lake nutrient criteria, in which FDEP proposed an alkalinity threshold of 50 mg/L as $CaCO_3$ to differentiate low and high alkalinity lakes (FDEP 2009).

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2.2.2 Data

Chl-*a* and selected water chemistry (alkalinity, color, nitrogen species, phosphorus species, pH, dissolved oxygen) data from Florida lakes were downloaded from the Florida IWR database, which comprises all of the STORET data for Florida (Florida DEP; http://publicfiles.dep.state.fl.us/dear/IWR/) (see *Appendix B1. Data Supporting EPA's Approach for Deriving Numeric Nutrient Criteria for Florida Lakes*). IWR/STORET data were augmented by FDEP with some of its own data not stored in the IWR database (FDEP 2009; see below). The IWR data set includes several years of monitoring by FDEP and other entities (public and private) in Florida.

The IWR run 39 (Florida DEP; http://publicfiles.dep.state.fl.us/dear/IWR/), included well over 300,000 individual observations from lakes. All data were spatially linked to USGS lake reach codes on the basis of station coordinates. FDEP also queried its own Laboratory Information Management System (LIMS) for QA information not provided in IWR. FDEP's data reduction and quality assurance (QA) on the data set consisted of the following steps:

- excluded uncorrected (total) chl-*a*. All values included were corrected chl-*a* values only;
- excluded values with a QA flag indicating problems with the reported value (e.g., sample collection problems, excess holding time, poor instrument calibration);
- replaced values reported as nondetects with ½ reported minimum detection limit (MDL; U. S. EPA 1998b);
- excluded invalid zero values (typically missing values miscoded as zero);
- calculated TN from NOx + total Kjeldahl nitrogen (TKN);
- spatially joined the stations to the National Hydrographic Database (NHD) on lake REACH;
- calculated daily average lake (REACH) TP, TN, color, alkalinity, and corrected chl-*a* where there were multiple samples on a given day; and
- lake color class was based on the long-term geometric mean (all observations).

EPA performed additional QA on the data obtained from FDEP consisting of the following steps:

- extreme values judged to be erroneous were removed: pH > 14; TN >100 mg/L;
- to remove estuarine waters, all sites with conductance >10,000 µS/cm were removed (several brackish water lakes with conductance < 10,000 µS/cm were retained);
- all data were log-transformed (natural log) except pH;
- all transformed concentration data were approximately normally distributed for use in model development (Appendix B-1);
- annual geometric means for each variable were calculated for each lake-year.

The data filters and QA resulted in a data set of 11,644 measurements of corrected chl-*a* in 1139 lakes. These comprised 1349 lake-years from 1996 to 2008. Some lake-years consisted of a single sample at a lake while others reflected multiple samples in a year.

To reduce short-term variability in nutrient and chl-*a* measurements and to obtain representative estimates of concentrations, EPA calculated annual geometric means from a minimum of four

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observations in a year, and required that both cool and warm seasons be represented in the annual geometric means. For the derivation of numeric nutrient criteria, EPA further screened the data set by the following:

- removing observations with chl-*a* < 0.25 µg/L;
- excluding lake-years with fewer than four measurements of TP, TN, and chl-*a*; and
- excluding years without at least one measurement in each season (TP, TN, and chl-*a*) - season 1: October to April; season 2: May to September.

Following these data reductions, the final data set for developing criteria consisted of 771 lake-years (annual geometric means of chl-*a*, TN, TP, and color) meeting the minimum observations requirements.

Log transformation

Water chemistry data were natural log (Ln) transformed prior to analysis to adhere to assumptions regarding normality required of many parametric statistical tests and to reduce the influence of extreme values typical of log-normally distributed nutrient concentration data on estimates of central tendency (e.g., Zar 1996). Figure 2-3, Figure 2-4, and Figure 2-5show quantile plots of Ln(TP), Ln(TN), and Ln (chl-*a*) indicating the approximation of a normal distribution provided by natural log transformations.

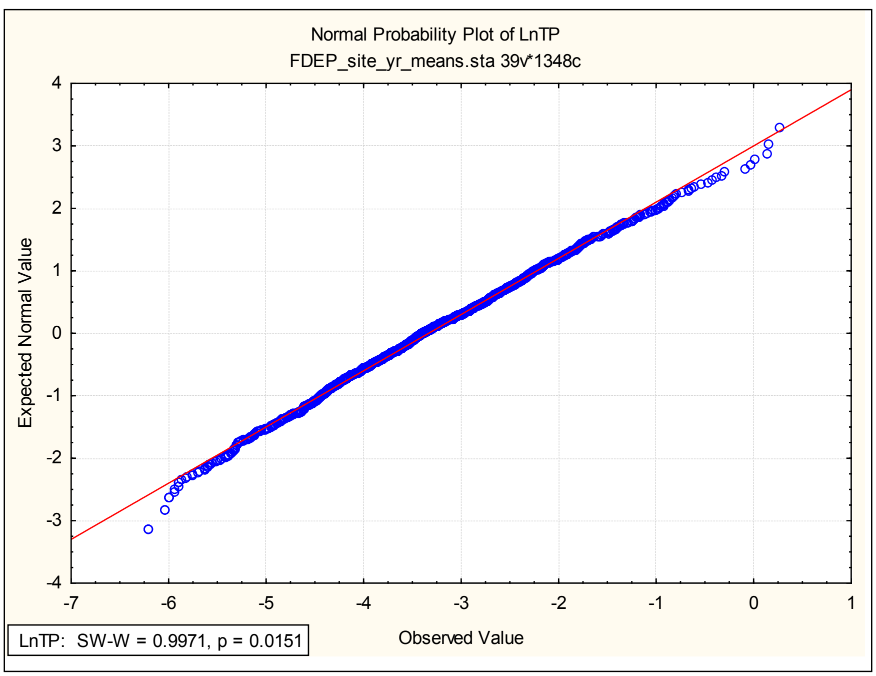

Figure 2-3. Normal probability plot of Ln(TP), annual geometric means.

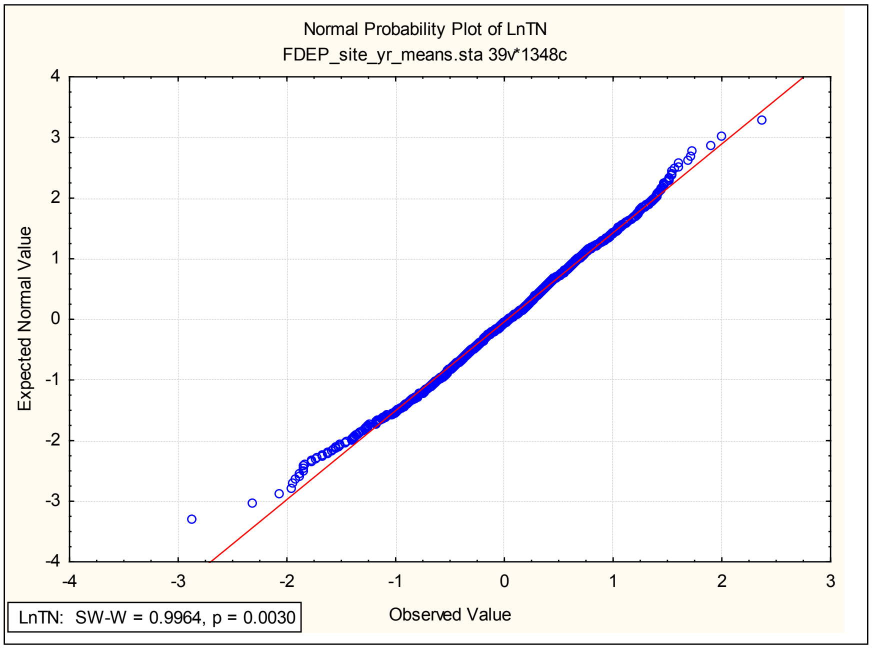

Figure 2-4. Normal probability plot of Ln(TN), annual geometric means

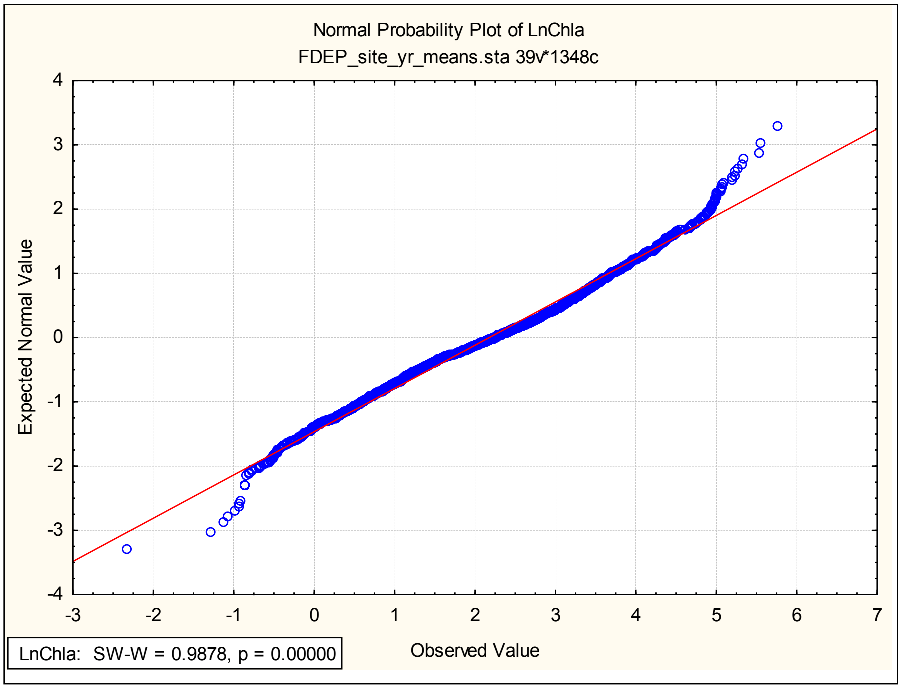

Figure 2-5. Normal probability plot of Ln(chl-*a*), annual geometric means.

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2.2.3 Classification Analysis

EPA examined the range of alkalinity in Florida lakes and found that the alkalinity of clear lakes (color ≤ 40 PCU) ranges from 0 (low alkalinity lakes with no ability to neutralize acid) to well over 200 mg/L total alkalinity (as $CaCO_3$) (Figure 2-6). Distribution of alkalinity in these lakes appears to be bimodal or trimodal, with peaks at 0.4 mg/L $CaCO_3$ (essentially 0 – low alkalinity lakes with little or no measurable alkalinity), moderately softwater lakes with a mode of approximately 5 to 6 mg/L, and alkaline lakes with a mode of approximately 50 mg/L. The boundary between the softwater and alkaline lakes is between 15 and 20 mg/L $CaCO_3$. Mean water color ranged from 2.5 PCU to more than 750 PCU. To facilitate examination, EPA divided mean color into the following 3 classes: clear (color ≤ 40 PCU); intermediate color (40–140 PCU), and dark color (>140 PCU). Based on classification and regression tree (CART) analysis, FDEP (2009) concluded that color break-points at 40 and 140 PCU best explained the distribution of color in Florida lakes, yielding three color classes: clear, intermediate color, and high color.

Examination of scatter plots of the Florida lake data revealed several relationships. First, TN, TP, and chl-*a* increase with alkalinity, except in the most highly colored lakes (>140 PCU) (Figure 2-7 through Figure 2-9). The slope of the relationship of nutrients and chl-*a* to alkalinity appears to increase sharply when total alkalinity is above 15–20 mg/L (as $CaCO_3$). In keeping with this observation, EPA set the threshold between low and high alkalinity lakes at 20 mg/L as $CaCO_3$.

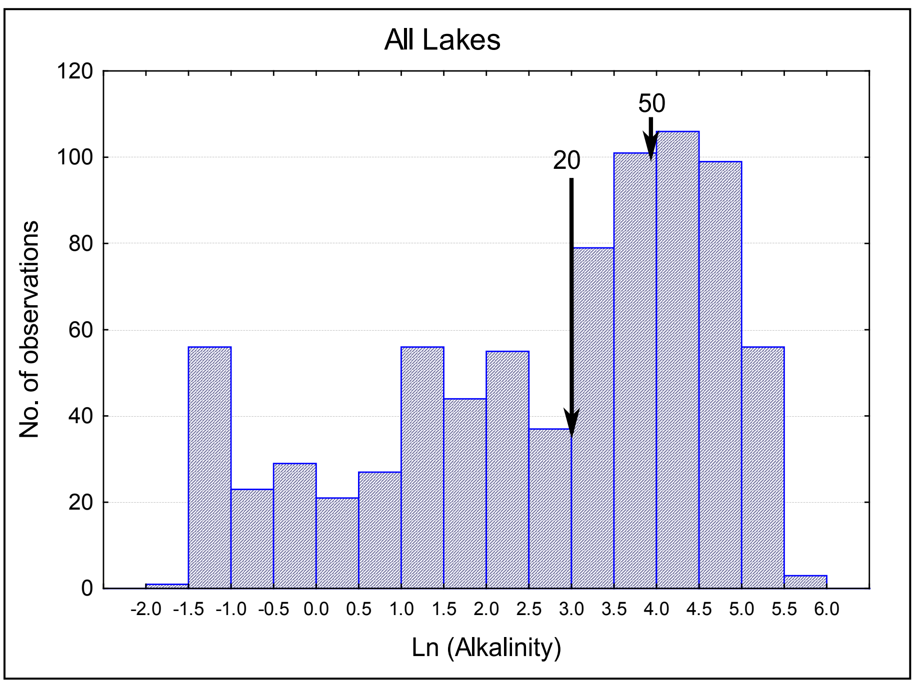

Figure 2-6. Distribution of average log alkalinity in Florida lakes (natural log scale).
Note that alkalinities of 20 and 50 mg/L shown by arrows; alkalinities are means of annual means for the period of record for each lake; N = 739 lakes.

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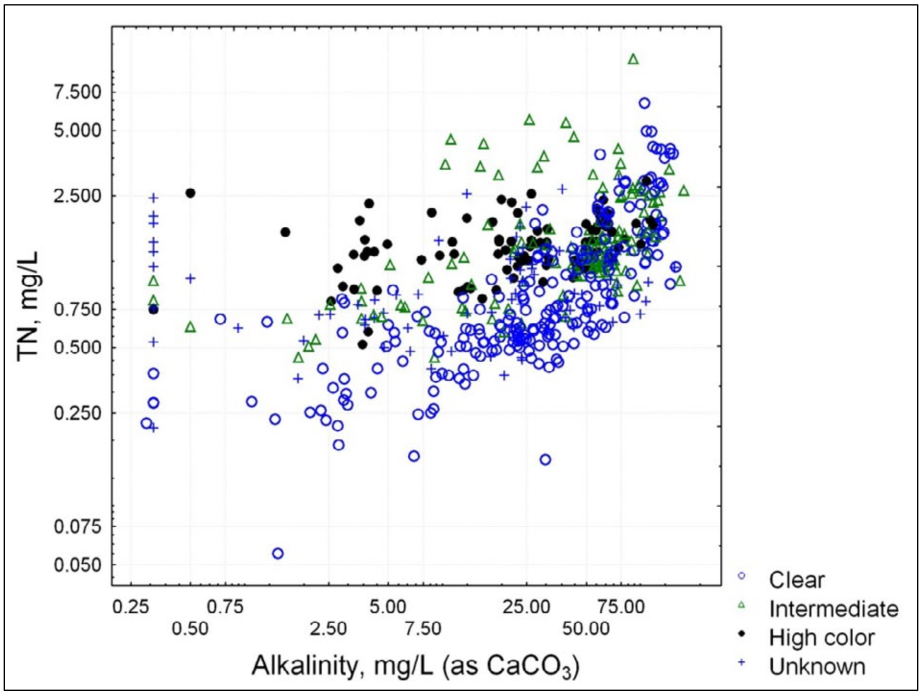

Figure 2-7. Scatter plot of TN and alkalinity in Florida lakes, annual geometric means, for 3 color classes.
Note that open circles: clear (≤ 40 PCU); triangles: intermediate color (40–140 PCU); closed circles: dark color (> 140 PCU); crosses: unknown color.

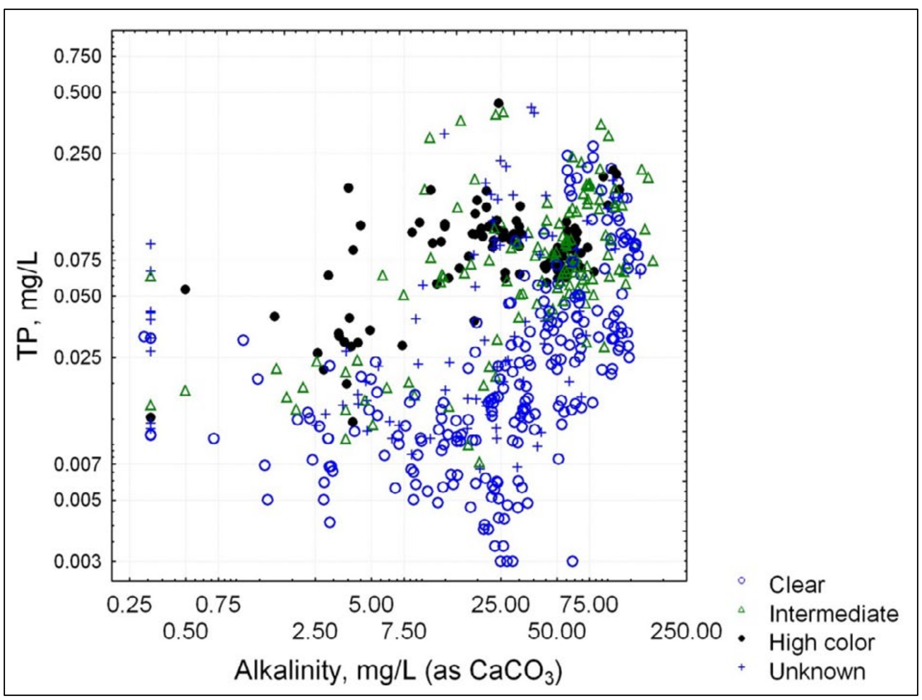

Figure 2-8. Scatter plot of TP and alkalinity in Florida lakes, annual geometric means.
Note that symbols are as in Figure 2-7.

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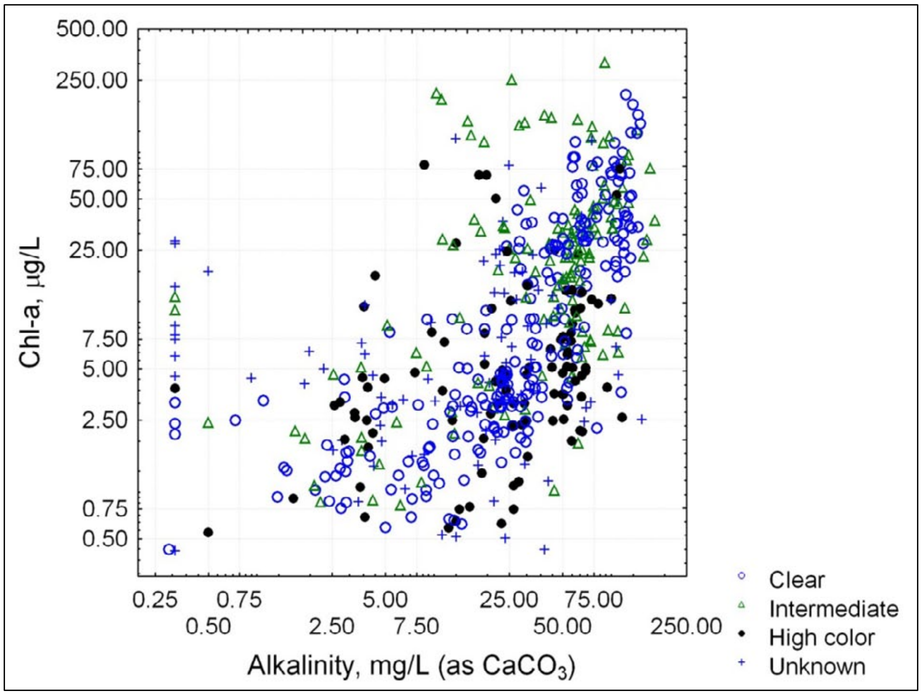

Figure 2-9. Scatter plot of chl-*a* and alkalinity in Florida lakes, annual geometric means.
Note that symbols are as in Figure 2-7.

Although TN and TP increase with color from clear lakes to lakes with colors of approximately 40 PCU, the association between TN, TP and color weakens above color > 40 PCU (Figure 2-10, Figure 2-11). Chl-*a* also increases up to the mid range (25–100 PCU) of color, and then declines at the highest color values (Figure 2-9). Nutrient and chl-*a* concentration appear to increase to a maximum between 25 to 75 PCU (Figure 2-10 through Figure 2-12). At very high color, nutrient concentrations are also high, but do not increase further with color. Chl-*a* concentration reaches a maximum in the range 25–75 PCU, and decline in colored lakes greater than 100 PCU. At the highest color values, algae are light-limited. Based on these observations and consideration of the color classes characterized by FDEP for Florida lakes (FDEP 2009), EPA classified Florida's lakes using the following color classes: ≤ 40 PCU, 40–140 PCU, and > 140 PCU.

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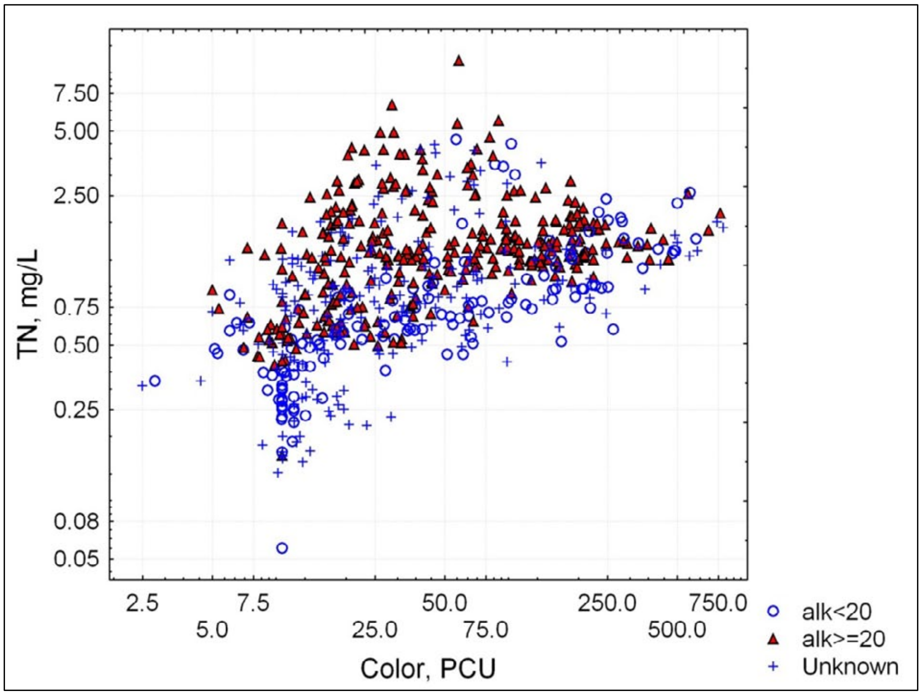

Figure 2-10. Scatter plot of TN and color in Florida lakes as annual geometric means for different lake alkalinity classes.
Note that open circles: alkalinity ≤ 20 $CaCO_3$ mg/L; triangles: alkalinity > 20 $CaCO_3$ mg/L; crosses: unknown alkalinity.

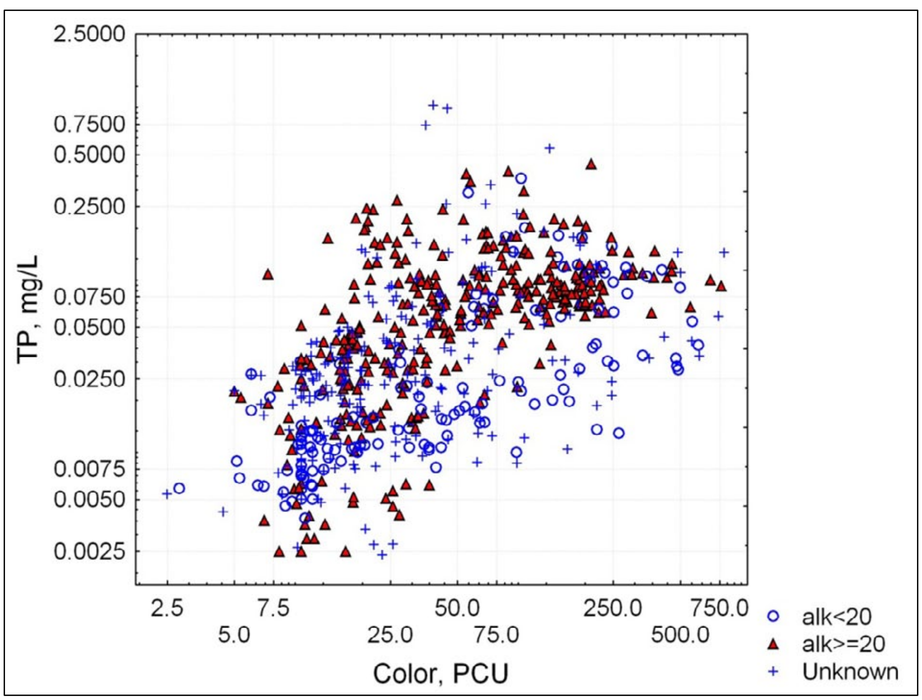

Figure 2-11. Scatter plot of TP and color in Florida lakes, annual geometric means for different lake alkalinity classes.
Note that symbols are as in Figure 2-10.

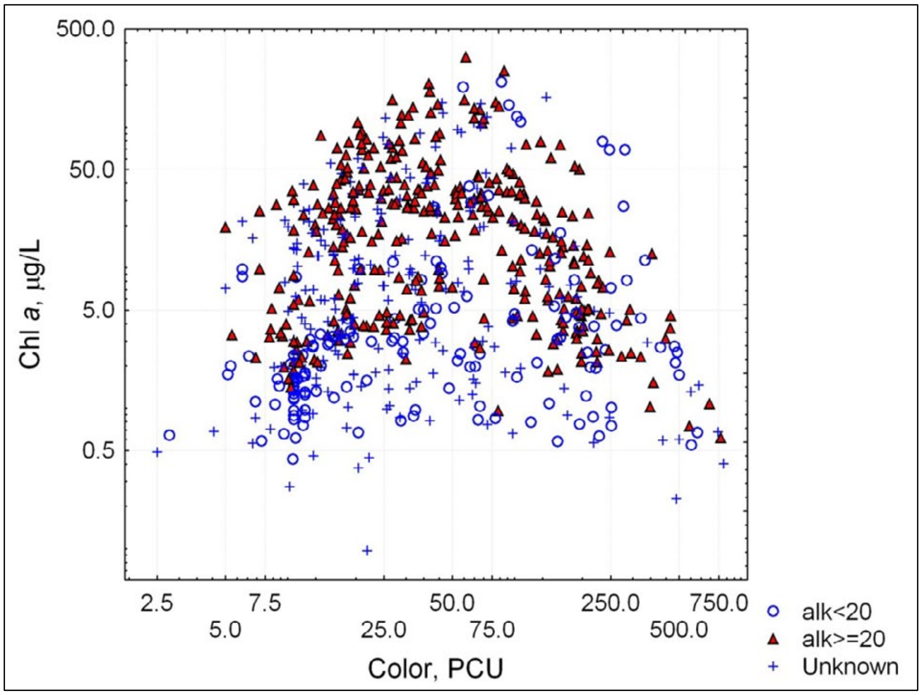

Figure 2-12. Scatter plot of chl-*a* and color in Florida lakes, annual geometric means for different lake alkalinity classes.
Note that symbols are as in Figure 2-10.

As a result of these analyses, EPA concluded that there is a scientific basis for classifying streams based on lake color and alkalinity. Furthermore, the analyses generally support FDEP's lake classification analyses from which it concluded that lakes could be classified according to the following color and alkalinity thresholds, as defined below (FDEP 2009):

- Moderately to highly colored lakes (color > 40 PCU)
- Clear lakes (color ≤ 40 PCU) with low alkalinity (≤ 50 mg/L $CaCO_3$)
- Clear lakes (color ≤ 40 PCU) with high alkalinity (> 50 mg/L $CaCO_3$)

In January 2010, EPA proposed a lake classification system similar to FDEP's classification system. However, based on the results (shown in Figure 2-6 through Figure 2-12), EPA concluded that the following classification system better accounts for the distributions of nutrients and the biological responses to nutrient enrichment:

- Moderately colored lakes (40 PCU < color ≤ 140 PCU)
- Highly colored lakes (color > 140 PCU)
- Clear lakes (color ≤ 40 PCU) with low alkalinity (≤ 20 mg/L $CaCO_3$)
- Clear lakes (color ≤ 40 PCU) with high alkalinity (> 20 mg/L $CaCO_3$)

The distribution of lakes across the State before and after data screening (described in Section 2.2.2) is shown in Figure 2-13 and Figure 2-14.

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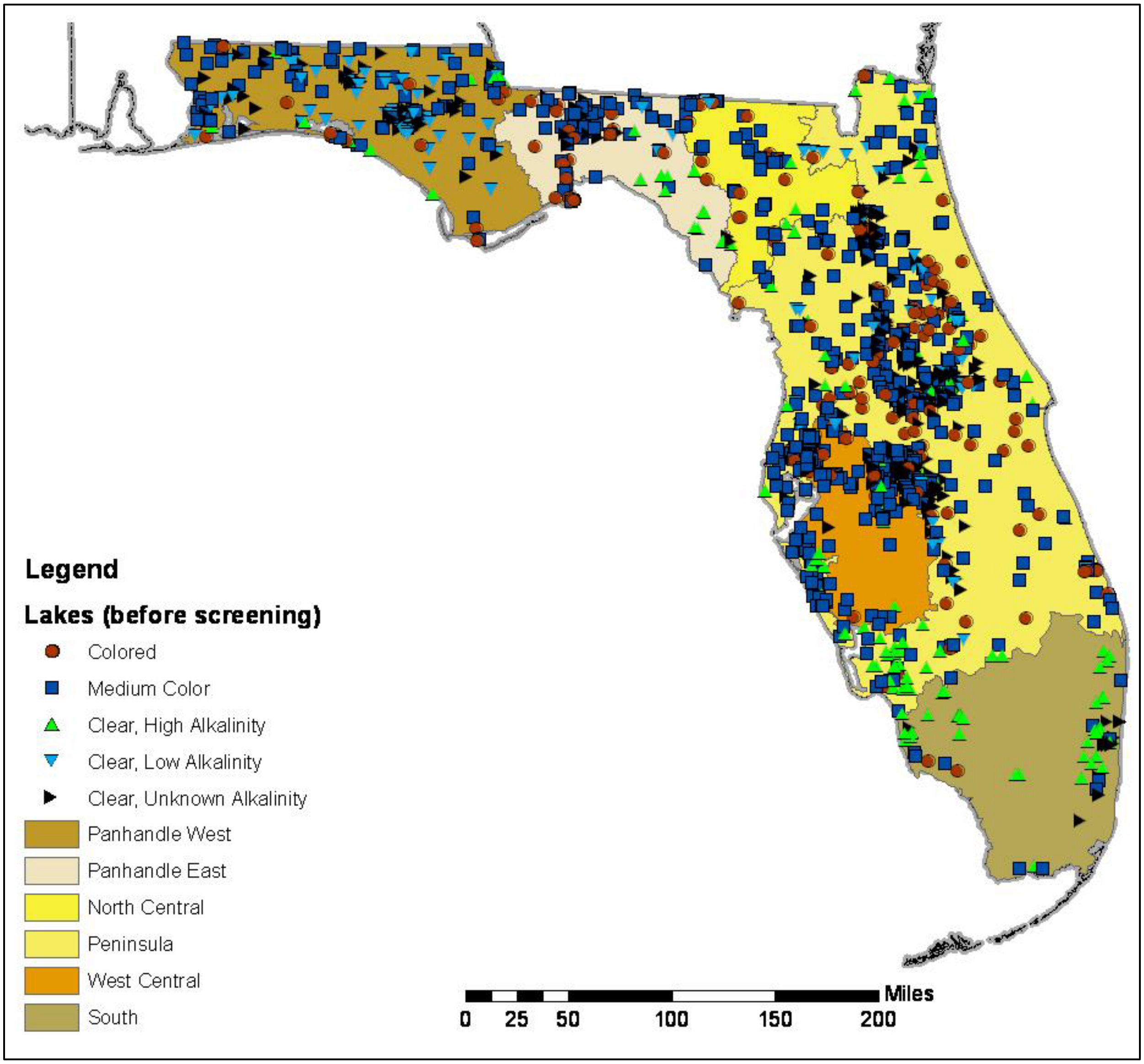

Figure 2-13. Distribution of Florida lakes by color and alkalinity class before data screening. Data screening methods are described in Section 2.2.2. EPA's stream Nutrient Watershed Regions (NWRs) are shown for geographical reference. See Chapter 1 for more information on stream NWRs.

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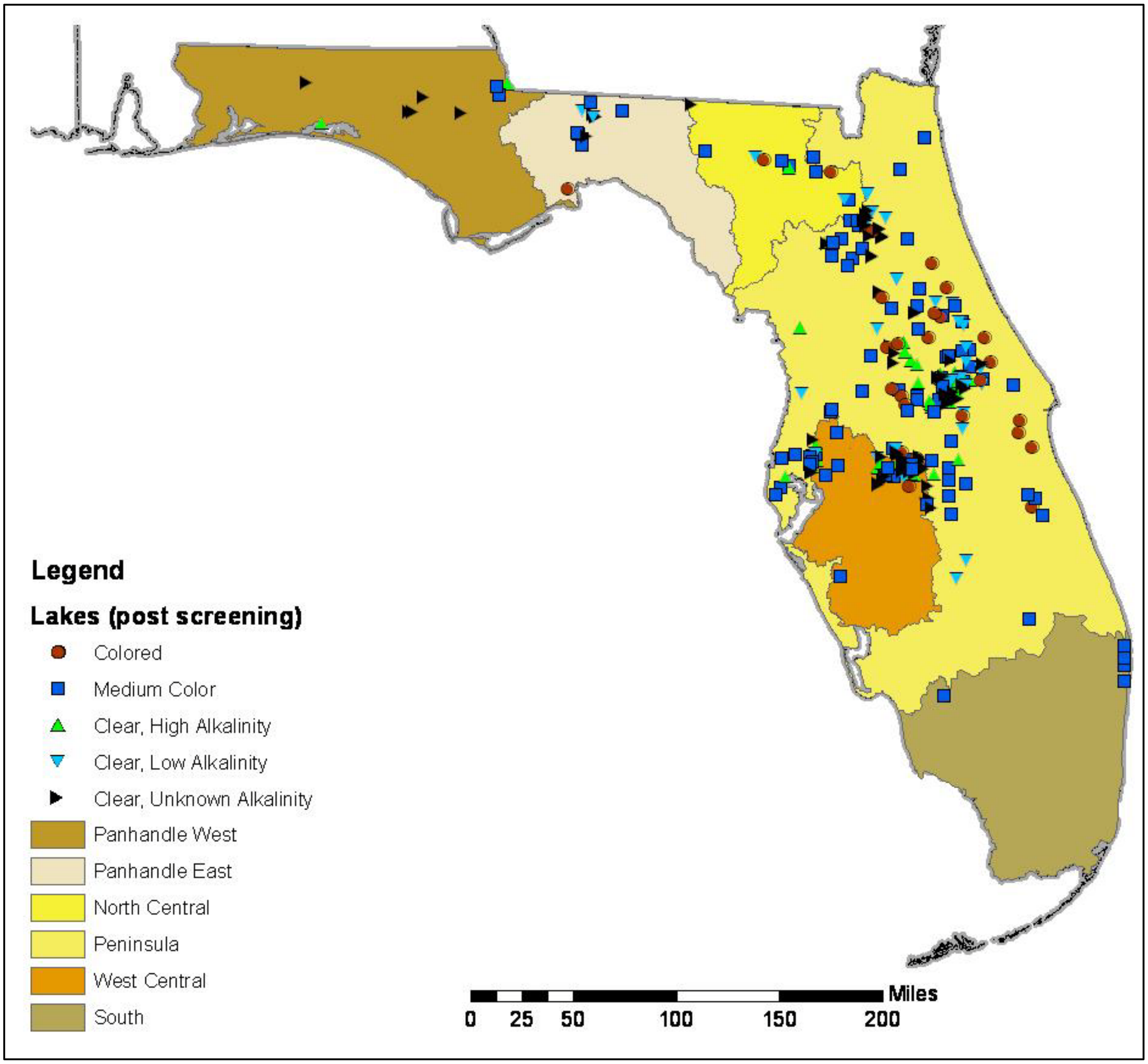

Figure 2-14. Distribution of Florida lakes by color and alkalinity class after data screening. Data screening methods are described in Section 2.2.2. EPA's stream Nutrient Watershed Regions (NWRs) are shown for geographical reference. See Chapter 1 for more information on stream NWRs.

EPA conducted additional statistical analyses as described below to ensure that a classification system to predict chl-*a* based on color and alkalinity is statistically supported, and compared its classification system to other candidate classifications.

Because alkalinity is not universally available in historical data, EPA also examined whether specific conductance can be used as a surrogate for alkalinity in classifying Florida lakes. In Florida's lake database, only a limited number of lakes (386) were measured for alkalinity while the majority of the lakes were measured for specific conductance. Because of its strong correlation with alkalinity, FDEP (2009) and EPA originally considered specific conductance as a surrogate for alkalinity. EPA evaluated the association between specific conductivity and alkalinity and concluded that alkalinity is a preferred parameter for lake classification because it

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is a more direct measure of the presence of carbonate rocks, such as limestone that are associated with natural elevated phosphorus levels. Changes in specific conductivity can be attributed to changes in alkalinity, but in many cases may be caused by increases in the concentrations of other compounds that originate from human activities (Herlihy et al. 1998). Thus, alkalinity is a more reliable indicator for characterizing natural background conditions for Florida lakes.

Classification of lakes into color and alkalinity classes requires that one characterize the long-term averages of these two lake characteristics. The number of samples required to compute a long-term average to accurately classify a particular lake depends strongly on the distance between the long-term average value and the threshold value between classes. For example, if the long-term average color for a particular lake is much less than or much greater than the threshold between classes (40 PCU), then one might only require one sample to accurately place a lake in the correct color class. Conversely, if the long-term average color is very near 40 PCU, then substantially more samples may be required.

Calculating long-term average color and alkalinity using all available data provides classifications that are more stable and less subject to change due to multi-year hydrological cycles. To identify cases in which estimates of long-term average color changed due to recent measurements, EPA examined the 82 lakes in which at last 40 color measurements were available. For each of these lakes, an initial lake classification was established by computing the geometric mean of the first 10 measurements. Then, changes in estimates of long-term average color over time were computed by adding one additional measurement to the sample and recalculating the geometric mean color. That is, long-term average color was computed for 11, 12, and 13 measurements, up to the end of the data record.

Of the 82 lakes for which data were available, color class changed in only two lakes over the period of record, so classification based on long-term average was stable for the vast majority of lakes in the data set. Color measurements and long-term average color for one of the lakes that did change classes is shown in Figure 2-15. In this lake, long-term average color was slightly higher than 40 PCU prior to 2006, and an extended period of low color shifted long-term average color below 40 PCU. For assessment of lakes exhibiting similar color dynamics, the causes for the recent period of persistently low color should be examined further to determine whether they reflect a cyclical trend in color or a permanent shift in lake class.

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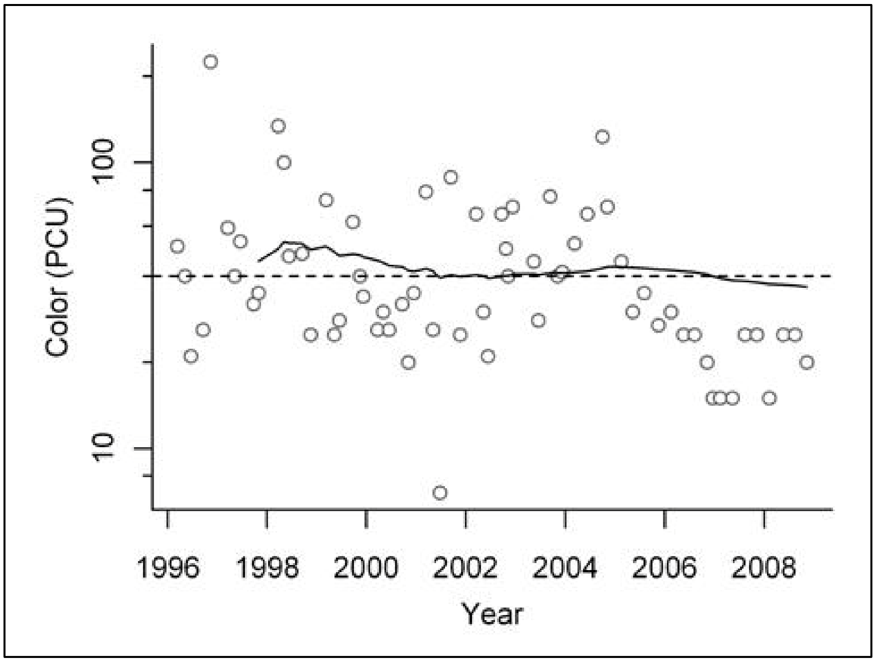

Figure 2-15. Color versus sampling time.
Long term average, based on all available data before indicated sample date, shown as solid line. Dashed line shows classification threshold of 40 PCU.

2.2.4 Statistical Analysis of Alternative Classification Systems

As discussed above, there are a variety of classification approaches and systems in use for Florida lakes. For numeric nutrient criteria derivation for Florida's lakes, EPA was primarily interested in the biological response, chl-*a*, as the principal factor to be explained by the classification system. EPA was also interested in the ability of the classification to support explanations of how nutrients affect the biological response. EPA analyzed lake water quality data (see Section 2.2.3) and found strong associations of TN, TP, and chl-*a* with color and alkalinity. EPA did not propose and is not finalizing criteria for water clarity, as Florida already has criteria for transparency and turbidity that are applicable to all Class I and III waters. These are expressed in terms of a measurable deviation from natural background (Rule 32-302.530(67) and (69), F.A.C.). EPA examined the strength of a classification system based on color and alkalinity through additional statistical analyses.

EPA was interested in identifying a classification scheme that resulted in nutrient stressor-response relationships that most accurately and precisely predicted chl-*a* concentrations. An R^2 value quantifies the proportion of variability in a response variable that a statistical model explains relative to an intercept-only null model; thus, when comparing different models applied to the same data set, a higher R^2 value corresponds to a model with more precise predictions. Therefore, EPA sought to maximize R^2 to identify the most appropriate lake classification system for developing numeric nutrient criteria and sought the most parsimonious model. EPA judged parsimony of various classification systems using Akaike's Information Criterion (AIC), which penalizes models according to the number of explanatory variables (e.g., Harrell 2001). Lower AIC values indicate more parsimonious models.

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Statistical analysis of various classification systems revealed that a combination of lake variables best predicted chl-*a* (Table 2-1). The classification system with the highest R^2 and lowest AIC for predicting chl-*a* was a continuous model of alkalinity, color, TN and TP (Table 2-1; Model 10):

$$chl-a = b_0 + b_1 TN + b_2 TP + b_3 Alkalinity + b_4 Color + b_5 Color^2$$

Although this continuous model was the best model with respect to R^2 and AIC, implementing criteria based on a continuous classification can be difficult for several reasons. First, each lake could potentially require a different chl-*a* criterion based on its natural characteristics of color and alkalinity. Second, inclusion of both TN and TP in the model defines an isopleth (where chl-*a* reaches its criterion) of TN and TP limits. Such isopleths have infinite numbers of permissible TN and TP combinations and assumes that both nutrients are co-limiting—an assumption that may not always hold true (e.g., light availability, carbon, temperature; micronutrients may also limit algal growth and biomass accumulation). EPA also notes that an infinite number of combinations of TN and TP criteria would likely present implementation challenges to the State. These challenges are similar to those that exist with Florida's existing narrative criteria, which require site-specific analysis that is resource intensive to implement. EPA's goal is to promulgate more generally applicable criteria. Therefore, EPA examined discrete classification schemes, namely, classifying lakes into a few alkalinity and color classes, and classifying lakes according to the approach proposed by Griffith et al. (1997) (Table 2-1; Models 5 and 6).

EPA found little difference between the Griffith et al. (1997) classification system (i.e., chl-*a* predicted by Lake Region) and EPA's lake models that predicted chl-*a* as a function of TN (and/or TP) + Lake Region (Table 2-1; Models 5 and 6) or as a function of TN (and/or TP) + 4 lake classes of color and alkalinity (Table 2-1; Models 13, 14, 19, 20). EPA evaluated two alternatives with four classes of lakes: the same slopes for the TN and TP relationship across the classes (Models 13, 14), and different slopes for the TN and TP across the classes (Models 19 and 20). The 4-class/same slope models were equivalent to the Lake Region models (cf. models 13, 14 and 5, 6), but allowing slopes to differ across classes yielded models with slightly higher R^2 and lower AIC (cf. models 19, 20 and 5, 6).

As a further evaluation of the Lake Region classification, EPA examined whether the seven lake regions identified as "sand hill lakes" differed from clear, low alkalinity, lakes that are not in regions identified as sand hill lake regions. Lakes in the seven sand hill regions were similar in all respects to clear, low alkalinity lakes in terms of their distributions of alkalinity, color, TN, TP and chl-*a*. Therefore, EPA concluded that the seven sand hill lake regions were redundant with the clear, low alkalinity class, and further—that the seven regions did not capture clear, low alkalinity lakes in other parts of the State. Thus, EPA concluded that this further lake region classification did not add predictive capability. Finally, existing data sets with corrected chl-*a* measurements included only 27 of the 47 lake regions (57%), leaving 43% of lake regions without data to support numeric nutrient criteria at the present time.

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Table 2-1. Model statistical summary (linear regression models).

Model	Description	Notes	Adj. R^2	AIC
1	CHL_A == f(TN)	Single nutrient only	0.527	1370
2	CHL_A = f(TP)		0.358	1514
3	CHL_A = f(TN + nutrient region)	Stream nutrient region model	0.562	1350
4	CHL_A = f(TP + nutrient region)		0.396	1493
5	CHL_A = f(TN + Lake Region)	Lake regions model (Griffith et al. 1997)	0.685	1195
6	CHL_A = f(TP + Lake Region)		0.653	1248
7	CHL_A = f(TN + TP + Lake Region)	Lake regions, both nutrients	0.697	1185
8	CHL_A= f(TN + alk + COLOR + $COLOR^2$ + PH)	Continuous classification (e.g., Lowe et al. 2009)	0.74	1090
9	CHL_A= f(TP + alk + COLOR + $COLOR^2$ + PH)		0.683	1184
10	CHL_A = f(TN + TP + alk + COLOR + $COLOR^2$)	Continuous, both nutrients in one model	0.756	1057
11	CHL_A= f(TN + clearlow + clearhi + color)	High and low alkalinity classes in clear streams (color ≤ 40), 1 class with color>40	0.588	1306
12	CHL_A= f(TP + clearlow + clearhi + color)		0.521	1378
13	CHL_A= f(TN + clearlow + clearhi + medcolor + hicolor)	3 color classes (≤ 40, 40-140, >140) and 2 alkalinity in the clear lakes. Same slope on nutrient variable (candidate EPA model)	0.689	1173
14	CHL_A= f(TP + clearlow + clearhi + medcolor + hicolor)		0.647	1233
15	CHL_A= f(TN + six color + alk categories)	3 color × 2 alkalinity classes, same slope on nutrient variable.	0.694	1167
16	CHL_A= f(TP + six color + alk categories)		0.650	1231
17	CHL_A = f(TN × six color + alk categories)	Slopes differ across classes, 3 color × 2 alkalinity classes.	0.718	1134
18	CHL_A = f(TP × six color + alk categories)		0.656	1228
19	CHL_A = f(TN × (clearlow + clearhi + medcolor + hicolor))	Slopes differ across classes, 4 classes of color and alkalinity (candidate EPA model)	0.715	1136
20	CHL_A = f(TP × (clearlow + clearhi + medcolor + hicolor))		0.653	1228

Note: all variables in the models were significant; N = 474 (most complete data set with all variables); variables enclosed within the f(.) are each represented in the model as the variable value multiplied by a constant regression coefficient.

EPA concluded that a four class classification system, based on color (40 and 140 PCU threshold) and alkalinity (20 mg/L $CaCO_3$ threshold), is scientifically-defensible and preferable to a classification system using continuous TN, TP, color, and alkalinity variables, or a classification system separating lakes into 47 different regions. The four lake classes are summarized in Table 2-2 (cross-reference: Table 2-1, Models 19, 20). The high color lake class, however, was not used for numeric nutrient criteria development (see Section 2.4.4, *Medium and high color lakes*).

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Table 2-2. EPA's lake classification for numeric nutrient criteria development.

Lake Class	Color Threshold	Alkalinity Threshold
High colored lakes*	>140 PCU	none
Medium colored lakes	40 -140 PCU	none
Clear, high alkalinity	≤ 40 PCU	> 20 mg/L $CaCO_3$
Clear, low alkalinity*	≤ 40 PCU	≤ 20 mg/L $CaCO_3$

Note: PCU – Platinum-Cobalt Units.

* These lake classes were important in model development, but were not employed for criteria development. See Section 2.4 and Table 2-6.

2.3 Methodology for Chl-*a* Criteria

Nearly all Florida lakes are shallow and include large areas of potential habitat for submerged and floating vascular plants. If the water is relatively clear, shallow zones of the lakes can be dominated by native submerged and floating plants. Natural primary production in such shallow lakes includes contributions from both submerged macrophytes and phytoplankton.

Algae and macrophytes in shallow lakes have complex dynamics under nutrient enrichment and physical disturbance conditions (Scheffer et al. 1993). Submerged macrophytes intercept nutrients for incorporation into plant biomass, where the nutrients are then unavailable to phytoplankton algae. Shallow, mesotrophic lakes with abundant macrophytes have clear water and relatively low chl-*a* concentrations. Such lakes are mesotrophic (intermediate level of productivity), but their water column chl-*a* concentrations are low because a large fraction of primary production is from the macrophytes and not phytoplankton algae. Such a stable state can be upset by (1) sudden loss of macrophytes because of disturbance (e.g., grass carp, hurricane, other disturbances); or (2) excessive nutrient loading leading to increased epiphytes and phytoplankton, which then shade the macrophytes (e.g., Bachmann et al. 1999; Lowe et al. 1999, 2001; Scheffer et al. 1993, 2001). Loss of the macrophytes can further increase TP concentrations (Bachmann et al. 2002), possibly from release from biomass and sediment, further increasing TP available to the phytoplankton. Such a new condition can also be stable, and turbidity from algae or resuspension of bottom sediment prevents macrophytes from reestablishing. Excess plant biomass problems can occur as dense algal blooms or a lake choked with invasive floating plants (e.g., *Hydrilla*, *Myriophyllum*, water hyacinth). The latter can also be in a stable state (Scheffer et al. 2003). Substantial evidence from both historical records and paleolimnology shows that several Florida lakes have transitioned from macrophyte-dominated, mesotrophic clear waters to phytoplankton-dominated, eutrophic (or hypereutrophic) turbid waters as the consequence of nutrient enrichment (Kenney et al. 2002; Lowe et al. 1999, 2001).

Submerged vegetation and phytoplankton support different consumer food webs. Macrophytes are the origin of a detritus- and periphyton-based food web, including macroinvertebrates, amphibians, fish, and birds. The phytoplankton food web supports zooplankton and fish. Greater numbers of fish are dependent on littoral, benthic macroinvertebrates for their diet than those that are dependent on zooplankton (e.g., Vadeboncoeur et al. 2002). In addition to supporting a food web, macrophytes also provide habitat and cover for fish (especially smaller forage fish and juvenile fish) and larger macroinvertebrates (e.g., Wetzel 1975). Vegetated areas of lakes have greater animal diversity (invertebrates, fish, birds) than open waters (Havens et al. 1996).

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Collapse of the submerged vegetation can result in an ecosystem change, specifically the reduction or elimination of native fish, invertebrate, and bird species dependent on the littoral habitat. Such effects are well-documented from eutrophication effects in the Great Lakes and elsewhere (e.g., Wetzel 1975; Winfield 2004). Thus, protecting aquatic life use requires maintenance of these system functions.

Because submerged vegetation is dependent on light, maintaining a lake's historic balance between algae and submerged plants requires maintaining overall historic water transparency. As noted previously, natural transparency varies widely in Florida lakes because of the range in water color from clear to deeply colored. Regardless of what the natural transparency of a lake might be, it is reduced by increased algal growth that results from anthropogenic nutrient enrichment. Accordingly, maintaining water column chl-*a* concentrations within traditional oligotrophic or mesotrophic limits reduces the risk that submerged vegetation and system functions will be unacceptably altered.

Because excess algal growth is associated with degradation in aquatic life and because chl-*a* concentrations are a measure of algal growth (biomass), EPA used chl-*a* concentrations as indicators of support for balanced natural populations of aquatic flora and fauna in each of the categories of Florida's lakes described above. EPA examined multiple lines of evidence to derive chl-*a* criteria that would be supportive of balanced natural populations of aquatic flora and fauna in Florida's lakes. These lines of evidence include trophic state of lakes, existing conditions, estimates of natural background conditions in Florida lakes (paleolimnological inferences, least-disturbed lakes, and model predictions), and user surveys. Different uncertainties exist in each of these lines of evidence, and EPA sought to balance these uncertainties when deriving the final chl-*a* criteria by qualitatively weighing the relative degree of uncertainty inherent to each line of evidence.

2.3.1 Trophic State

Lakes are typically classified into the following five trophic classes to reflect nutrient conditions and overall productivity: ultra-oligotrophic, oligotrophic, mesotrophic, eutrophic, and hypertrophic. The Organisation for Economic Development and Co-Operation (OECD) trophic state classification (OECD 1982) has been internationally accepted for nearly 30 years, including acceptance by both EPA and FDEP (FDEP 2009; U.S. EPA 2000). Table 2-3 shows these trophic classes, associated TP and chl-*a* values, and consensus assessment of use impairment. Anthropogenically caused eutrophic conditions are generally acknowledged to be undesirable for drinking water, recreation, and aquatic life uses (e.g., U.S. EPA 2000).

Table 2-3. OECD (1982) and Salas and Martino (1991) trophic categories.

	OECD (1982)			Salas and Martino (1991)	
Trophic Category	TP (mg/L)	Chl-*a* (µg/L)	Use Impairment	TP (mg/L)	Use Impairment
Ultra-oligotrophic	< 0.004	< 1.0			Low
Oligotrophic	< 0.010	< 2.5	Little	< 0.028	Low
Mesotrophic	0.010-0.035	2.5–8	Variable	0.028–0.070	Variable
Eutrophic	0.035–0.10	8–25	Great	> 0.070	High
Hypertrophic	>0.10	>25			Very High

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FDEP has historically used a modified version of Carlson's Trophic State Index (TSI) to manage lakes throughout the State (FDEP 2009). Note that while TSI is not a direct measure of any single lake characteristic, it is a convenient index for relating transparency, TN, TP, and chl-*a* on a common scale that can also be related to trophic state categories. A TSI based on TP and a TSI based on chl-*a* are only as predictive of each other as the original regressions used to develop them. In this final rule, EPA used trophic state and chl-*a* concentrations to define use thresholds, as originally determined by OECD (1982), and subsequently modified by FDEP (2009) and EPA's analysis (Table 2-3). Criteria based on TP and TN are then derived from the response relationships of chl-*a* to the nutrients. EPA used chl-*a* instead of the TSI to define aquatic life and use thresholds because it directly relates to support of Florida's designated uses (through primary productivity, see Figure 2-2).

Salas and Martino (1991) analyzed tropical and subtropical lakes of the Americas (including lakes in north Florida) to identify trophic states corresponding to TP concentrations. Based on lake managers' identification of the trophic categories (similar procedure as used by OECD), they concluded that tropical-subtropical TP concentrations were higher than the equivalent temperate concentrations by a factor of approximately two (Table 2-3). Based on the geographic distribution of the data used by Salas and Martino, and observed concentrations of TP in Florida lakes, FDEP argued (2009) that the subtropical lakes of Florida are more similar to the tropical and subtropical lakes analyzed by Salas and Martino (1991) than to the lakes analyzed in OECD (1982).

Because Salas and Martino (1991) defined different trophic states by TP concentrations, EPA used regression relationships between TP and chl-*a* in clear lakes to convert the threshold TP concentrations provided by Salas and Martino to comparable chl-*a* concentrations (Figure 2-16). The resulting chl-*a* threshold between oligotrophic and mesotrophic in clear lakes could be estimated as the point at which the probability density was the same for both oligotrophic and mesotrophic classes. This threshold was approximately 10 μg/L. A similar threshold between mesotrophic and eutrophic conditions was identified at approximately 27 μg/L. Using the same approach for colored lakes yielded a threshold between mesotrophic and eutrophic conditions of 14 μg/L.

Sources of uncertainty in these threshold values include the following: (1) Salas and Martino (1991) used a set of tropical and subtropical lakes to identify different trophic states, and none of the lakes in Florida are located in a tropical climate. Therefore, the Salas and Martino definitions of trophic classes may not be exactly applicable to Florida lakes, (2) uncertainty exists in the relationships used to convert TP to chl-*a* concentrations, and (3) other thresholds between trophic classes could potentially be defined from the probability density functions.

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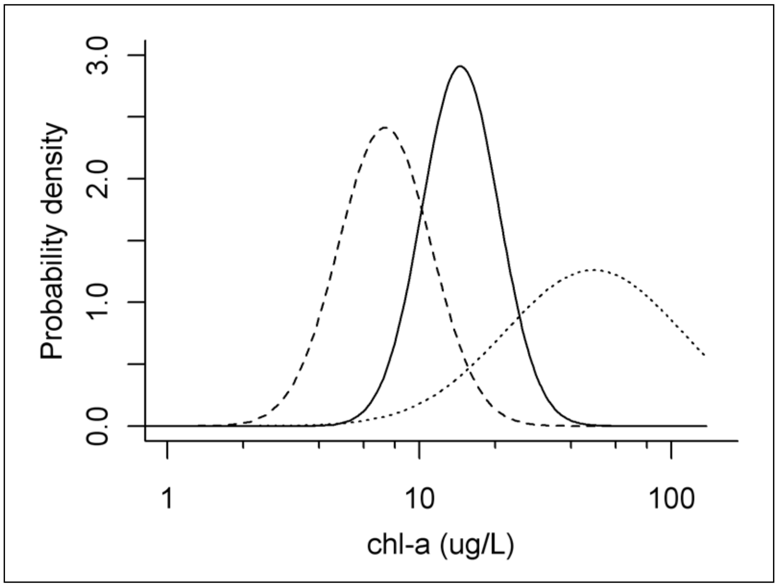

Figure 2-16. Probability density distributions of chl-*a* in different trophic classes for clear lakes, based on Salas and Martino (1991).
Note that dashed line: oligotrophic, solid line: mesotrophic, dotted line: eutrophic.

2.3.2 Existing Conditions

Considering the currently existing distributions of chl-*a* concentrations in different lake classes provides another means of assessing potential chl-*a* criteria. In clear, low alkalinity Florida lakes, the 25th and 75th percentile of the distribution of existing chl-*a* concentrations were 2 and 7 µg/L. In clear, high alkalinity lakes, the 25th and 75th percentile were 4 and 25 µg/L, and in colored lakes the 25th and 75th percentile were 2 and 18 µg/L (Figure 2-17).

Uncertainty in defining the characteristics of existing lake conditions arises primarily from the issue of whether the lakes for which data are available provide a representative sample of lakes in a particular class. Data used to estimate existing conditions were not gathered with a randomized sample; however, a qualitative evaluation of the spatial distribution of lakes indicates that they are reasonably representative. A second source of uncertainty in using existing conditions to inform criteria is the relative degree of human disturbance in different classes of lakes. By selecting a low percentile (e.g., 25th percentile) of the distribution of existing conditions, one typically attempts to approximate a high percentile of minimally-disturbed conditions, but this percentile selection depends on the relative degree of disturbance present in existing conditions. Less disturbed existing conditions warrant higher percentiles. EPA qualitatively considered the relative degree of human disturbance in each of the lake classes when evaluating this line of evidence for criteria derivation.

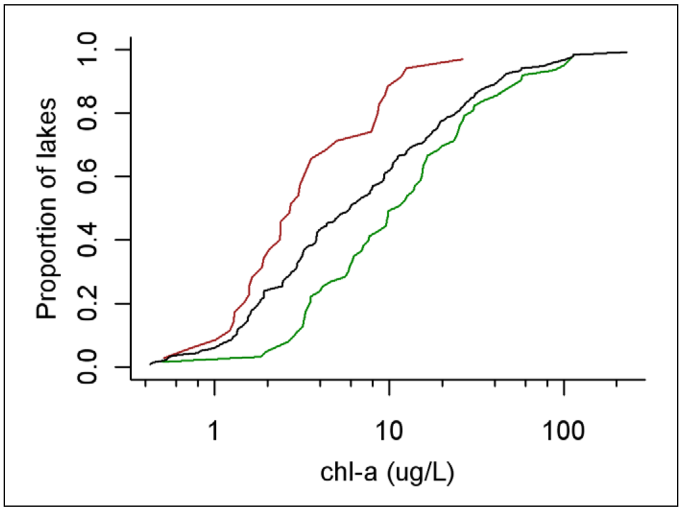

Figure 2-17. Empirical cumulative distributions of currently existing chl-*a* concentrations in Florida lakes.
Note that brown line: clear, low alkalinity lakes; green line: clear, high alkalinity lakes; black line: colored lakes.

2.3.3 Paleolimnological Studies

Paleolimnological studies in Florida lakes provide an estimate of chl-*a* concentrations prior to extensive human disturbance by inferring chl-*a* concentrations from the skeletal remains of diatom communities in deep sediment cores (Whitmore and Brenner 2002). Paleolimnological studies conducted at Lakes Shipp, Lulu, Haines, May, Conine and Bonny in the Florida peninsula suggest that the average chl-*a* concentrations in these lakes ranged historically between 14 and 20 µg/L. Analyses of deep sediment cores from two other lakes (Lakes Wauberg and Hancock) indicated that historic chl-*a* in these lakes ranged historically from 38–48 µg/L and 74–133 µg/L, respectively. See *Appendix B2. Analysis of Paleolimnological Data.* Although the sediments of several Florida lakes have been analyzed in paleolimnological studies (e.g., Riedinger-Whitmore et al. 2005; Whitmore and Brenner 2002), the total number of lakes analyzed is not sufficient to develop reliable estimates of historical background conditions statewide. However, the results indicate that more than 50% of lakes studied increased in TP concentration over the past century.

2.3.4 Least-Disturbed Lakes

Least-disturbed lakes were identified by FDEP based on surrounding land use, and these lakes were used to derive candidate chl-*a* criteria (Paul and Gerritsen 2003). In general, chl-*a* concentrations in least-disturbed lakes were less than those observed in all lakes (Figure 2-18), but the differences between least-disturbed and all lakes distributions varied by lake class. In particular, the distribution of chl-*a* concentrations in clear, high alkalinity lakes was substantially lower than observed in the all lakes distribution. However, the small sample size in this category (n=8) makes it difficult to determine whether this distribution is fully characteristic of these types of least-disturbed lakes. In least-disturbed, clear, low alkalinity lakes, the 75th and 90th percentiles of the chl-*a* distribution were 5 and 8 µg/L, respectively. For clear alkaline lakes, the same two percentiles were 7 and 10 µg/L, and for colored lakes, the 75th and 90th percentiles were 13 and 30 µg/L.

Uncertainties in the characterization of least-disturbed lakes arise primarily from the criteria used to identify these lakes. The amount of disturbance in these lakes, relative to a minimally-disturbed condition, could not be determined, and uncertainty in the quality of these lakes was considered when weighing this line of evidence.

2.3.5 Model Predictions of Natural Background Concentrations

EPA used information from a model developed by the St. Johns River Water Management District (Morphoedaphic Index [MEI] model, Lowe et al. 2009; *Appendix B1. Data Supporting EPA's Approach for Deriving Numeric Nutrient Criteria for Florida Lakes*) that predicts natural background chl-*a* and TP concentrations for a lake given its depth, alkalinity, and color. The MEI can be used to predict historical background conditions for any lake. The MEI is based on the observation that limestone bedrock, the natural source of background phosphorus, also imparts increased alkalinity. Thus, the MEI, calculated as lake alkalinity divided by lake depth, is a strong predictor for natural phosphorus concentrations (Vighi and Chiaudani 1985). Because of the additional influence of color on lake chl-*a* concentrations, the model for chl-*a* used both MEI and color as predictor variables. In Florida, the MEI model was calibrated using a set of lakes in which human activities in the catchment were minimal. The MEI was found to be significantly related to chl-*a* concentrations (Figure 2-19). However, because substantial predictive uncertainty was observed, particularly at low values of MEI, these predictions did not contribute strongly to final criterion derivation.

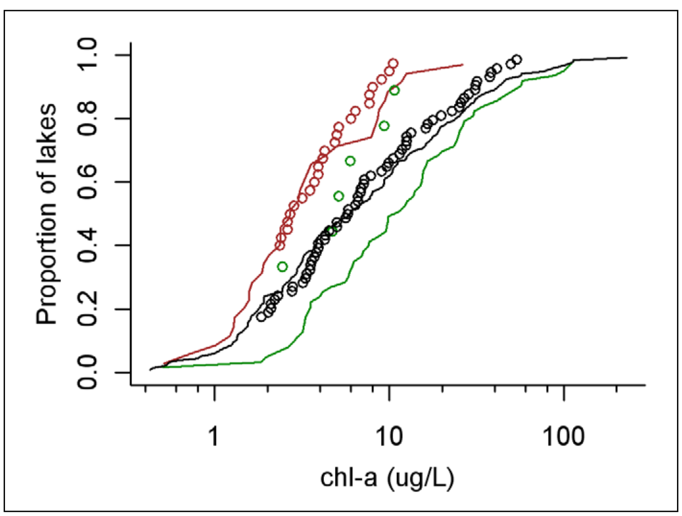

Figure 2-18. Chl-*a* distributions in least-disturbed lakes (from Paul and Gerritsen 2003). Note that lines shown existing distribution of chl-*a* (see Figure 2-17) and symbols show distribution of least-disturbed lakes for each lake class. Brown: clear, low alkalinity lakes, green: clear, high alkalinity lakes, black: colored lakes. Also note that the lower values of each distribution not shown because chl-*a* values were assigned values of 1.0 µg/L due to detection limits.

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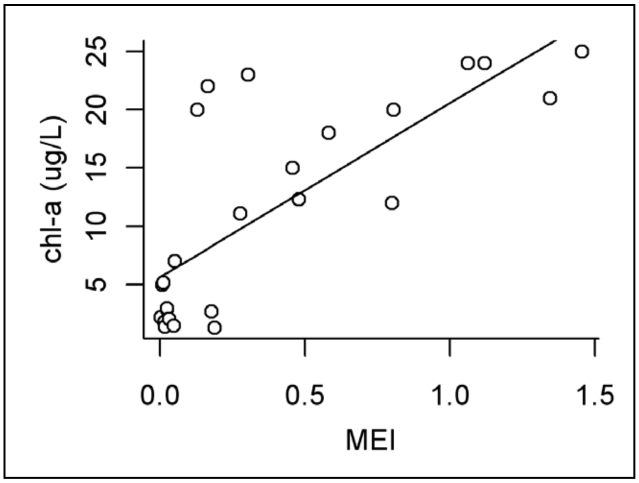

Figure 2-19. MEI versus chl-*a* in calibration data set (from Lowe et al. 2009).
Note that the straight line is the simple linear regression fit; lake color did not contribute significantly to the model.

The MEI model was applied to alkalinity and color data available from Florida lakes, and summarized by lake class (Figure 2-20). The 75th percentiles of these distributions were clear, low alkalinity lakes: 6.8– 7.3 µg/L; clear, high alkalinity lakes: 15–17 µg/L; and colored lakes: 13–17 µg/L.

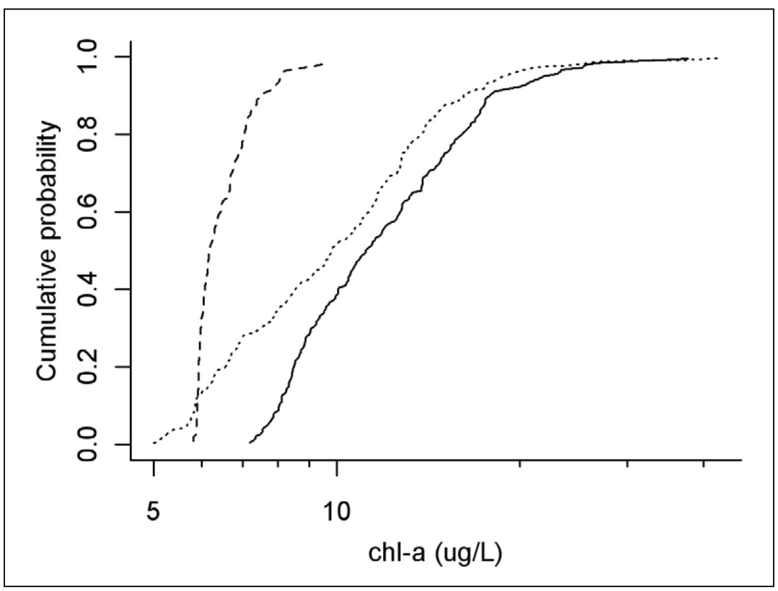

Figure 2-20. MEI predictions of chl-*a* distributions in different lake classes.
Note that the dashed line: clear, low alkalinity lakes; solid line: clear, high alkalinity lakes; dotted line: colored lakes.

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2.3.6 User Perceptions

A user perception study conducted in Florida demonstrated that there were differences in user perceptions differed depending upon lake region (Hoyer et al. 2004). In that study, when lake users responded to a question concerning suitability of the lake for recreation and aesthetic enjoyment by saying, "beautiful, could not be nicer," chl-*a* ranged from approximately 30 µg/L in the Central Valley Lake Region (generally high color) to approximately 3 µg/L in the Trail Ridge Region (generally uncolored lakes). These studies only associated chl-*a* concentrations with the perception of whether recreation in and on the lake was desirable, and did not directly assess support for aquatic life use. Although the range in chl-*a* from this study spans an order of magnitude, the chl-*a* concentrations are informative as boundaries associated with recreational use and aesthetics, which are a component of Florida's Class III designated use.

2.3.7 Summary and Final Criteria

EPA based the final numeric nutrient criteria for lakes upon multiple lines of evidence, which are presented in Table 2-4. The rationale for each lake class's chl-*a* criterion is detailed below. The primary lines of evidence used to derive chl-*a* criteria were trophic status, existing distributions of chl-*a* concentrations in all sampled lakes, and distributions of chl-*a* concentrations in least-disturbed lakes. Small samples sizes and uncertainties in the paleolimnological studies and in the model predictions limited the degree to which these lines of evidence could be considered, and they were considered secondarily. Similarly, as noted above, user perceptions did not directly inform aquatic life use support, and were therefore only considered secondarily.

Clear, low alkalinity lakes

Three lines of evidence informed the final derivation of a chl-*a* criterion concentration. First, EPA judged that the appropriate trophic state of clear, low alkalinity lakes is oligotrophic, and therefore, the work of Salas and Martino suggested a chl-*a* criterion of 10 µg/L. Least-disturbed lakes provide a second line of evidence, suggesting criteria ranging from 5 – 8 µg/L, depending on whether one selects the 75th or 90th percentile of the distribution. Finally, examination of the distribution of chl-*a* concentrations in all sampled lakes of this class indicated that 25th and 75th percentiles of existing chl-*a* concentrations were 2 and 7 µg/L. Current EPA guidance suggests that a low percentile of values collected from all lakes or a high percentile of values collected from least-disturbed reference lakes provide appropriate criteria (U.S. EPA 2000). As discussed above, EPA was uncertain regarding the degree to which the least-disturbed lakes represented minimally-disturbed conditions. With minimally disturbed lakes, the 90th percentile would be appropriate (as applied with the streams nutrient criteria), but uncertainty in the selection of these lakes suggested a criterion somewhat less than the 90th percentile value. Conversely, the similarity between the existing lakes and the least-disturbed distributions suggests that a higher proportion of all sampled lakes in this class are least-disturbed than is typically observed. Hence, a relatively high percentile of the existing lakes distribution was appropriate. Finally, the boundary between mesotropic and oligotrophic conditions was higher than values derived from either least-disturbed or existing lakes, but uncertainties with regard to the applicability of these numbers to Florida (as discussed above) led EPA to weight this line of evidence less strongly than the other two. Therefore, EPA adjusted down from the 90th percentile value of least-disturbed lakes because of uncertainty regarding the condition of these lakes, and adjusted substantially up from the 25th percentile of all sampled lakes because chl-*a* concentrations in existing lakes were comparable to those in least-disturbed lakes. These considerations led EPA to finalize a criterion value of 6 µg/L.

This lake type—clear, low alkalinity, ultra-oligotrophic, subtropical—is not known to occur in other states. The naturally-occurring fauna of these ultra-oligotrophic lakes is unique because they have a large number of pollution-sensitive insects normally found in very clean water streams, such as clubtail dragonflies (family *Gomphidae*), caddisflies of the families *Leptoceridae* and *Polycentropodidae*, and mayflies of the family *Leptophlebiidae* (Gerritsen et al. 2000). These clear, low alkalinity lakes have more of these sensitive, indicator organisms than any other class of Florida lakes (Gerritsen et al. 2000). They also have unique macroinvertebrate communities unlike any other lakes in Florida; currently undocumented fish and amphibian communities, and currently undocumented phytoplankton and submerged vegetation communities uniquely adapted to the extreme oligotrophy (e.g., Johnson and Castenholz 2000).

Clear, high alkalinity lakes

Because natural geological sources increase TP concentrations in these lakes, EPA determined that the appropriate trophic state of this class of lakes is mesotrophic. Hence, one potential criterion value was the boundary between mesotrophic and eutrophic chl-*a* concentrations, at 27 µg/L. Only a limited number of least-disturbed lakes were available in this lake class (n=8), and so this line of evidence was uncertain and not weighted strongly for this class of lakes. Instead, this line of evidence helped EPA interpret the existing all-sampled lake distribution. As with clear, low alkalinity lakes, the upper percentiles of the least-disturbed distribution were higher than the 25th percentile of chl-*a* concentrations in all sampled existing lakes, suggesting that criteria should be based on a higher percentile than the recommended 25th percentile. However, in these clear, high alkalinity lakes, the distribution of least-disturbed chl-*a* concentrations was not as similar to the all lakes distribution as was observed in clear, low alkalinity lakes (see Figure 2-18), and so, a smaller upward adjustment from the 25th percentile was warranted. Based on these considerations, EPA selected a final criterion of 20 µg/L.

Colored lakes

In general, EPA expected that colored lakes would maintain a mesotrophic status, which, for colored lakes, suggested chl-*a* criterion of 14 µg/L. However, uncertainty associated with this line of evidence was somewhat higher than in the other lake classes because TP accounted for less variability in chl-*a* concentrations in colored lakes (Table 2-6). Furthermore, chl-*a* concentrations across all sampled lakes and least-disturbed colored lakes spanned a wide range of values (Figure 2-12), as particularly manifested by the 90th percentile value of the least-disturbed distribution (30 µg/L). Therefore, to balance between the possibility that naturally high chl-*a* concentrations are present in some lakes, and the expected trophic status for the majority of lakes, EPA finalized 20 µg/L as the criterion concentration.

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Table 2-4. Summary of chl-*a* concentrations presented across different lines of evidence.

Line of Evidence	Lake Classes and Range of Chl-*a* (µg/L)		
	Clear, Low Alkalinity	Clear, High Alkalinity	Colored
Trophic state	10	27	14
Existing concentrations (25th and 75th %tiles)	2–7	4–25	2–18
Paleolimnology	—	—	20
Least-disturbed lakes (75th and 90th %tiles)	5–8	7–10	13–30
Morphoedaphic Index (75th and 90th %tiles)	7–17	15–17	13–17
Lake user perceptions*	3	—	30

Note: lake user perceptions is highlighted with an asterisk because it does not reflect a designated use of aquatic life support.

2.4 Methodology for Deriving Numeric TN and TP Criteria in Lakes

Based on the classification analysis previously described, numeric TN and TP criteria were based on regression relationships estimated within discrete classes defined by color thresholds at 40 and 140 PCU and an alkalinity threshold at 20 mg/L $CaCO_3$. Consideration and comparison of nutrient stressor-response relationships within classes defined by these thresholds further refined these classifications. Numeric nutrient criteria were derived from regression estimates of relationships between TN, TP, and chl-*a*.

2.4.1 Further Classification Considerations

The following three different discrete classifications based on the established color and alkalinity thresholds were considered:

- three categories (clear, low alkalinity; clear high alkalinity; color > 40 PCU);
- four categories (clear, low alkalinity; clear high alkalinity; 40 PCU < color ≤ 140 PCU; color > 140 PCU); and
- six categories (three color categories multiplies by two alkalinity categories).

For these models, EPA examined the effect of independent slopes on TN and TP among the categories vs. common slopes for all categories. The four category model (including a high color category) better accounted for variability in chl-*a* than the three category model. The six category model further improved on the four category model, but some of the categories contained insufficient numbers of samples to support an independent model within that category. Models based on independent slopes yielded higher R^2 values and lower AIC values than pooled slope models (see Table 2-1).

EPA also considered classifying lakes based on seasonality in the State. Analysis of lakes with paired summer and winter observations showed that there is a tendency for summer chl-*a* values to be higher. The analysis used all lake years that met the data criteria for the regression analysis: a minimum of four samples per year, with at least one in each season. The data set was the same

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as that used for regression analysis to derive criteria. Each lake-year thus had a geometric mean of cool and warm season observations. Seasonal differences were tested statistically with the paired t-test.

The mean seasonal difference increased with color, from negligible in clear lakes to almost double in high-color lakes (Table 2-5). Intermediate-color lakes had 40% higher chl-*a* in summer. An increase in productivity in summer is to be expected. The seasonal difference is statistically significant due to the large sample size, but it is small and negligible in the context of overall variability among lakes. Trophic state is a long-term average attribute, and not something that changes seasonally. Seasonal differences in chl-*a* concentrations are taken into account by the annual geometric mean, and by the requirement that samples are distributed throughout the year. The magnitude, frequency and duration components of the criteria protect the long-term average trophic state of a lake. Because the objective of the criteria is to protect the long-term trophic state, EPA concluded that the differences in clear and moderately colored lakes were not sufficient to derive seasonally adjusted criteria.

Table 2-5. Geometric mean chl-*a* concentrations in cool and warm seasons.

Lake type	N (lake-years)	Chl-*a*, µg/L	
		Cool season	Warm season
Clear (all)	417	7.0	7.8
Clear, low alkalinity	107	1.9	2.2
Clear, high alkalinity	179	12.2	14.4
Intermediate color	254	12.0	17.0
High color	96	2.5	4.8

2.4.2 Regression Models

EPA developed bivariate regressions of chl-*a* on TN and TP, respectively, for four categories of alkalinity and color, and estimated separate slopes for each category (Table 2-6). The regressions for clear, high alkalinity lakes and for medium color lakes accounted for a large proportion of variability in chl-*a* values, but regressions for clear, low alkalinity lakes and for high color lakes were relatively imprecise.

Clear, low alkalinity lakes can have extremely low chl-*a* concentrations, as well as very low TN and TP (filled blue circles in Figure 2-21, Figure 2-22). The very low chl-*a* in some lakes may be due in part to carbon limitation at extremely low alkalinities (acid lakes) (e.g., Barnese and Schelske 1994; Fairchild and Sherman 1993; Hein 1997). In addition, the very low chl-*a* and nutrient values in these lakes might be at or below the detection limits for chl-*a* or for TP. Detection limits varied somewhat depending upon the person or organization that collected the data, and these differences would increase the observed variance in chl-*a* and nutrient values at low values.

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In general, the clear, low alkalinity lakes appear to be part of the same relationship as the clear, high alkalinity lakes (filled red triangles in Figure 2-21, Figure 2-22). However, observations from low alkalinity lakes were more widely distributed about the mean regression line. The nutrient-chl-*a* relationships in clear, low alkalinity lakes and clear, high alkalinity lakes were qualitatively similar. The all clear lakes regression relationship (solid lines in Figure 2-21, Figure 2-22) was statistically similar to relationships estimated for clear, high alkalinity lakes (dashed lines in Figure 2-21, Figure 2-22). Figure 2-21 and Figure 2-22 illustrate how the clear, low alkalinity lakes appear to be part of the same relationship as the clear, high alkalinity lakes; however, the low alkalinity lakes are somewhat more spread out from the regression line, possibly due to detection limits or to carbon limitation in acidic lakes. To avoid confounding by detection limits and by potential carbon limitation in acidic lakes, the all clear lakes regressions were used to derive criteria for all clear lakes ($R^2_{TP} = 0.65$; $R^2_{TN} = 0.76$; Table 2-6).

Scatterplots showed that high color lakes typically have low chl-*a* concentrations, with only very few mean chl-*a* values above the criterion value of 20 µg/L (open triangles in Figure 2-23, Figure 2-24). These low chl-*a* concentrations are most likely the result of light limitation of algal growth by the dark color in these lakes. Thus, the highly colored lakes were pooled with the intermediate color lakes for application of criteria, as explained in Section 2.4.3 below and shown in Table 2-6.

Table 2-6. Chl-*a* vs. TN, TP regression results.

Lake Class	N	TN		TP	
		Formula y = Ln(chl-*a*) x = Ln(TN)	R^2 (adj)	Formula y = Ln(chl-*a*) x = Ln(TP)	R^2 (adj)
Clear, low alkalinity	107	y = 0.58 x + 1.23	0.15	y = 0.76 x + 4.12	0.30
Clear, high alkalinity	179	y = 1.77 x + 2.43	0.81	y = 0.86 x + 5.57	0.61
All clear lakes*	417	y = 1.67 x + 2.42	0.76	y = 1.11 x + 6.26	0.65
Intermediate color*	254	y = 2.05 x + 1.93	0.66	y = 1.11 x + 5.68	0.57
High color	96	N.S.	N.S.	y = 0.47 x + 2.49	0.06

* Regressions used for criteria. See accompanying text.

N.S. = not statistically-significant

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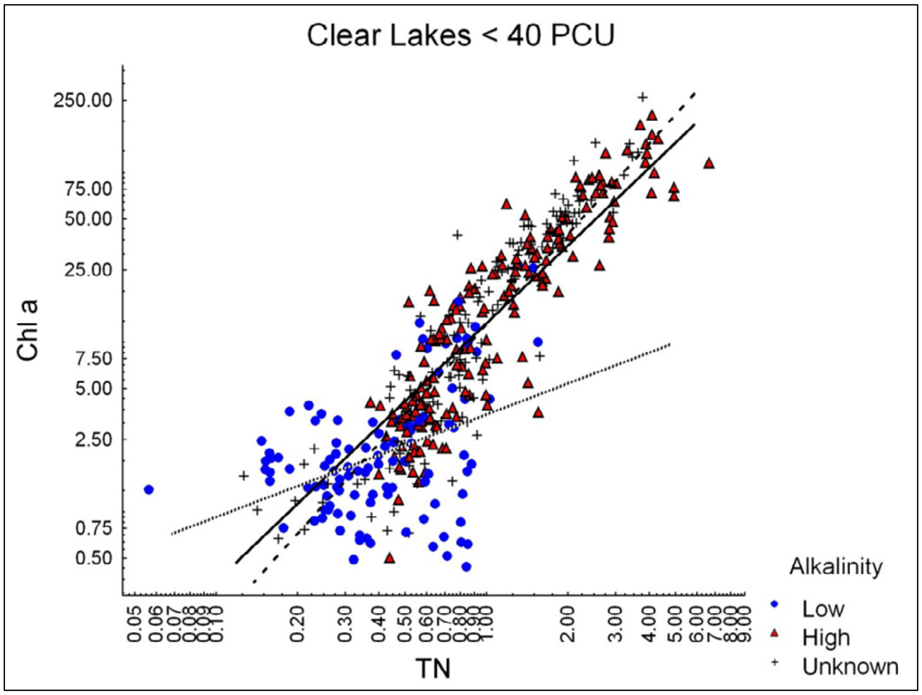

Figure 2-21. Chl-*a* – TN regressions for clear lakes.
Note that triangles: high alkalinity; closed circles: low alkalinity; crosses: unknown. Dotted line: low alkalinity regression line; dashed line: high alkalinity regression; solid line: all clear lakes. Annual geometric means.

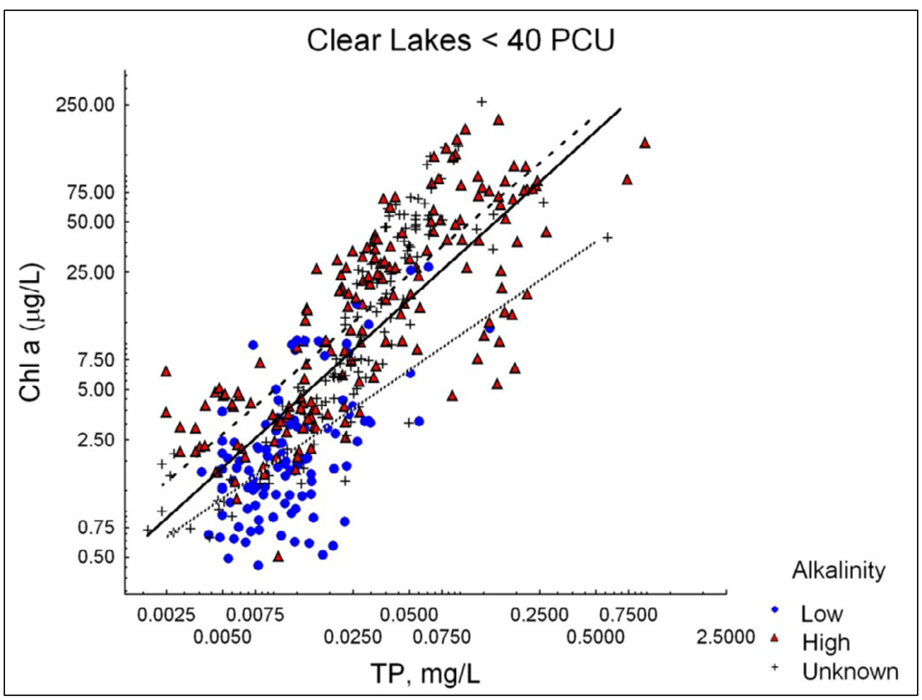

Figure 2-22. Chl-*a* – TP regressions for clear lakes.
Note that symbols are as in Figure 2-21.

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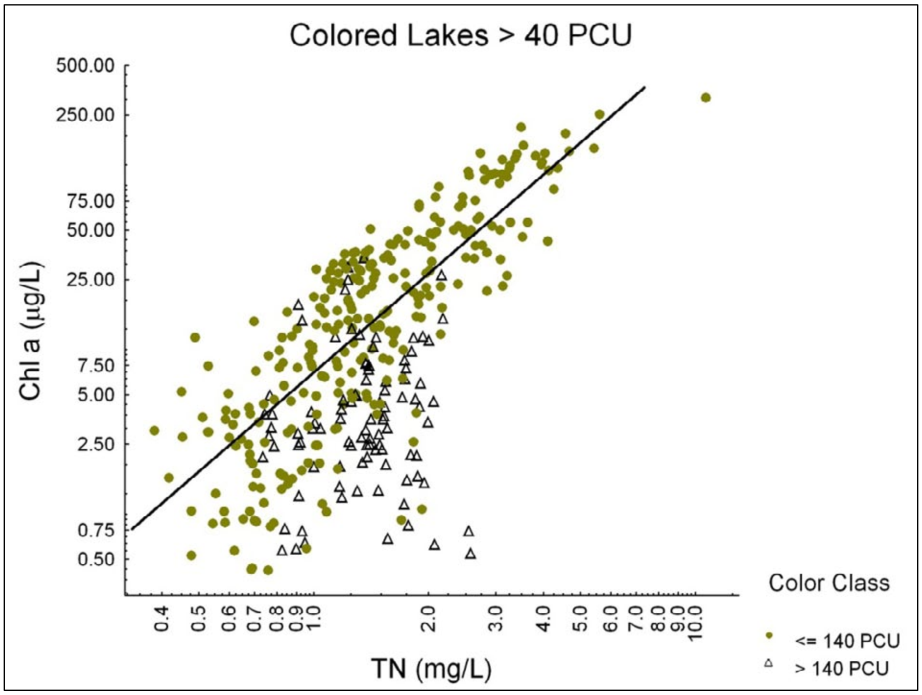

Figure 2-23. Chl-*a* – TN regression in intermediate colored lakes.
Note that closed circles: intermediate color lakes; open triangles: high color lakes. Regression line is for medium colored lakes; high color lakes NS.

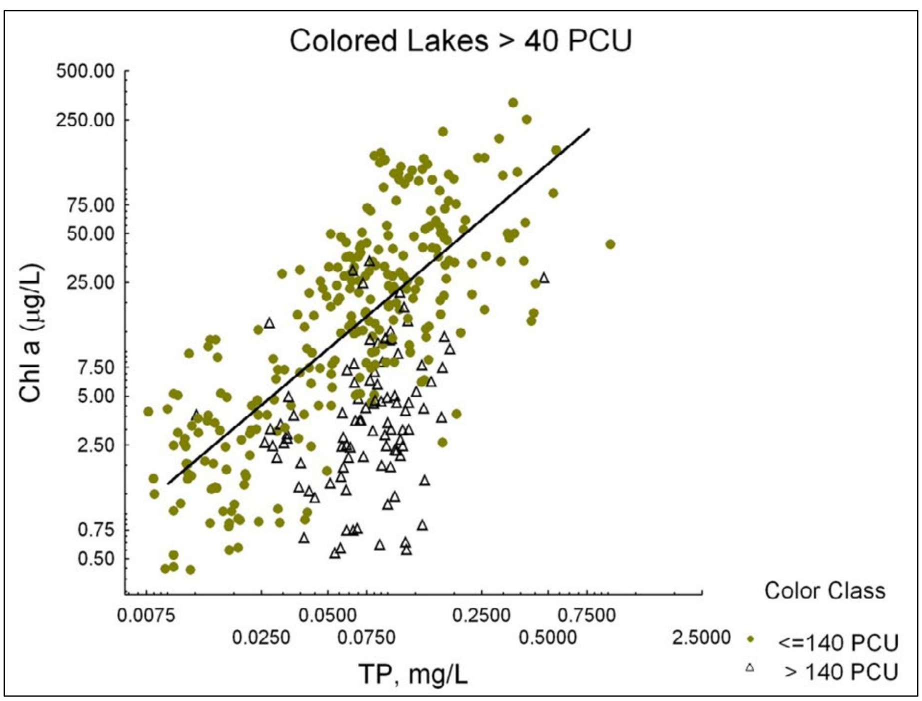

Figure 2-24. Chl-*a* – TP regression in intermediate colored lakes.
Note that symbols are as in Figure 2-23.

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Residuals

EPA examined the distribution of the residuals from the four regressions in shown in Figure 2-21, Figure 2-22, Figure 2-23, and Figure 2-24. The residuals plotted close to a straight line in probability plots, indicating no practical departures from normality (Figure 2-25).

TN-TP Correlation

Criteria for TN and TP were determined with simple linear regression models using each of TN and TP as independent variables. TN and TP are strongly correlated in the data set, which can potentially influence the accuracy of simple linear regression estimates of relationships between chl-*a* and either one of these nutrients. However, the estimated TN-chl a and TP-chl-*a* represent limiting cases that appropriately inform criteria derivation. More specifically, the estimated TN-chl-*a* relationship is an accurate representation of the nutrient stressor-response relationship if one assumes that the majority of lakes in Florida are nitrogen-limited, whereas the TP-chl-*a* relationship is an accurate estimate if one assumes that the majority of lakes are phosphorus-limited. The proportion of lakes that are nitrogen- or phosphorus-limited is not known, and indeed, the nature of nutrient limitation can shift over time within a particular lake. Hence, adoption of criteria based on the two limiting cases described above provides appropriately protective criteria.

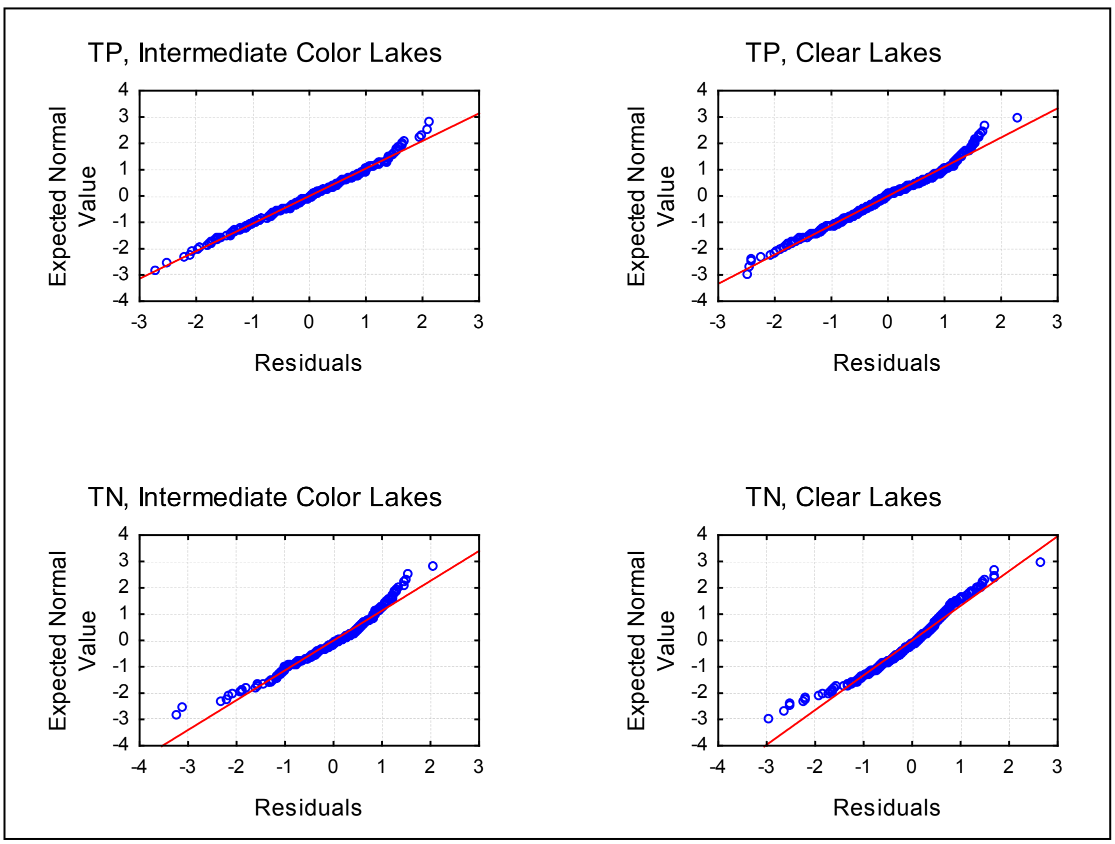

Figure 2-25. Quantile plots of residuals from chl-*a* – nutrient regressions used to develop criteria. Note that the residuals did not show constant variance and the magnitude of the variance was larger at the lower end of the predicted values range (Figure 2-26). As discussed previously, these larger residual variances may be due to the effect of censored data (detection limits) at the low values, or to other limitations of chl-*a* (e.g., carbon limitation in the clear, low alkalinity lakes).

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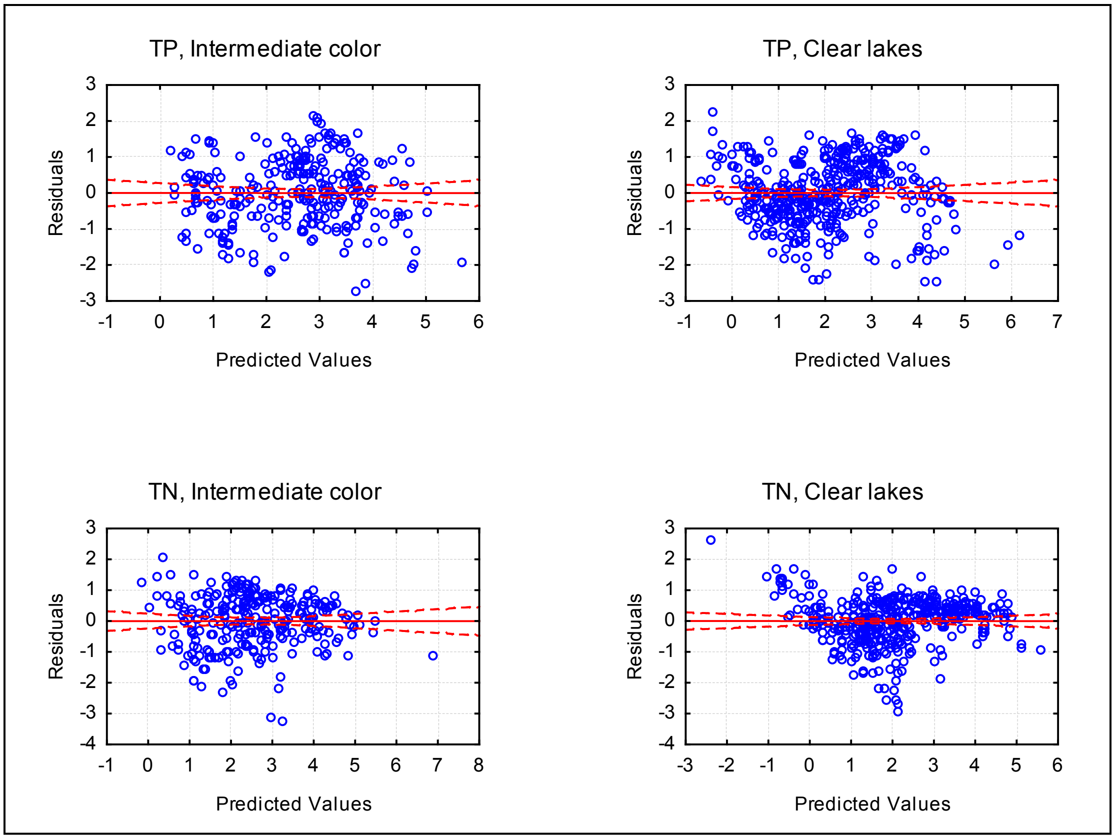

Figure 2-26. Residuals vs. predicted values from chl-*a* – nutrient regressions used to develop criteria.

In the classification exercise, EPA determined that the best predictive model for chl-*a* was a linear multiple regression model where chl a was a function of TN, TP, alkalinity, color, and (color).[2] Because the single regressions are not the same as the best "explanatory" model, we expect that residuals analysis will show that there are other potential variables that could contribute to the regression (i.e., there is more variability of the chlorophyll response that could be explained by additional explanatory variables). This was indeed the case, as R^2 values for the multiple regression models were somewhat higher than those observed for single variable regressions. However, the differences between multiple and single variable regression models were not substantial.

Regression residuals were normally distributed, but showed unequal variance, as well as potential over- or under-prediction because of the simplified regression models. The areas of strongest over- and under-prediction are candidates for development of site-specific alternative criteria.

2.4.3 Other Regression Modeling Approaches

EPA considered inverse regression (using chl-*a* as predictive variable and nutrients as response), but inverse regression does not address the questions that are relevant to deriving nutrient criteria. The objective of criteria development is not to estimate the nutrient concentration that yields chl-*a* of 20 µg/L, but is rather to derive nutrient concentrations in the context of the probabilities of chl-*a* exceeding 20 µg/L (e.g., 25% and 75%). In other words, the distribution of chl-*a* is the variable of interest. The baseline and upper nutrient values of the criteria are intended to reflect the probability of observed chl-*a* exceeding its criterion value, 25% and 75% probability, respectively. Such probability of chl-*a* values exceeding a given value are estimated by the 50% prediction intervals.

EPA also considered simultaneously modeling the effects of TN and TP on chl-*a* for each of the discrete classes. Including both TN and TP improved the precision of the nutrient stressor-response models. However, these models introduce complications with regard to implementing the criteria because TN and TP criteria are interdependent. That is, the TN criterion value one would derive from the stressor-response relationship would depend on a TP value that would have to be pre-specified, or the TP criterion would depend on a value of TN that would have to be pre-specified.

Measurement error in nutrient concentrations

Ordinary least squares (OLS) regression models assume that the explanatory variables are measured without error. This assumption is not possible in any real data set (e.g., see Zar 1996)—so the practical assumption is that errors in explanatory variables are negligible compared to the response variables. The effect of measurement error in the *x* variable is to expand the observed range of *x* values, which effectively reduces the magnitude of estimates of the regression slope. The degree to which measurement error attenuates an estimated regression relationship is a function of the measurement error of the *x* variable and the standard deviation of the "error-free" *x* variable. In this case, the effect of measurement error relative to total standard deviation is reduced because EPA used annual means estimated from a minimum of four samples, rather than individual measurements. Furthermore, the measurement errors in the annual averages used to estimate regression relationships are the same as would be observed when assessing different lakes for their compliance with the criteria. Thus, the estimated regression relationships provide the appropriate predictive relationship for setting criteria.

Evaluation of model accuracy

The accuracy of the regression models was evaluated by considering the degree to which other possible confounding variables (namely, lake color and alkalinity) were correlated with TN and TP, and by considering how lake color and alkalinity influenced the estimated relationship within each class.

Splitting the data set into colored and clear lakes greatly reduced the strength of correlation between lake color and nutrient concentrations in the colored lake class (Table 2-7, Table 2-8). However, correlation between color and nutrient concentration remained strong in the clear lake class, and correlation between alkalinity and nutrient concentrations remained strong in both classes.

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Table 2-7. Correlation coefficients between listed covariate and Ln(TP) in the full data set, and after splitting data set into clear and colored lakes.

	Full Data Set	Clear	Colored
Ln(alkalinity)	0.59	0.60	0.57
Ln(color)	0.50	0.52	0.20

Table 2-8. Correlation coefficients between listed covariate and Ln(TN) in the full data set, and after splitting data set into clear and colored lakes.

	Full Data Set	Clear	Colored
Ln(alkalinity)	0.52	0.76	0.47
Ln(color)	0.60	0.60	0.15

Evaluation of model precision

Regression models computed within each lake class yielded a range of criterion values that would potentially be applicable for different lakes within each class. This range of values is explicitly included in the final rule as modified TN and TP criteria (see below). Therefore, the final rule effectively accounts for the uncertainty in the predictive relationships between nutrient concentrations and chl-*a*.

To further evaluate the potential effects of co-varying variables on the accuracy of estimated relationships, the data within each lake class was split further by alkalinity and by color, and regression relationships estimated with each subclass. The results of further sub-classification by alkalinity can be seen in Table 2-6, Figure 2-21, and Figure 2-22. Based on these analyses, EPA concluded that the effects of alkalinity on the estimated relationships between chl-*a* and nutrients were small, and therefore the estimated relationships were deemed to be accurate.

Similarly, estimates of regression relationships with further subclasses of color (within the existing colored and clear classes) did not have strong effects on the estimated nutrient-chl-*a* relationships (see Figure 2-27 for one example).

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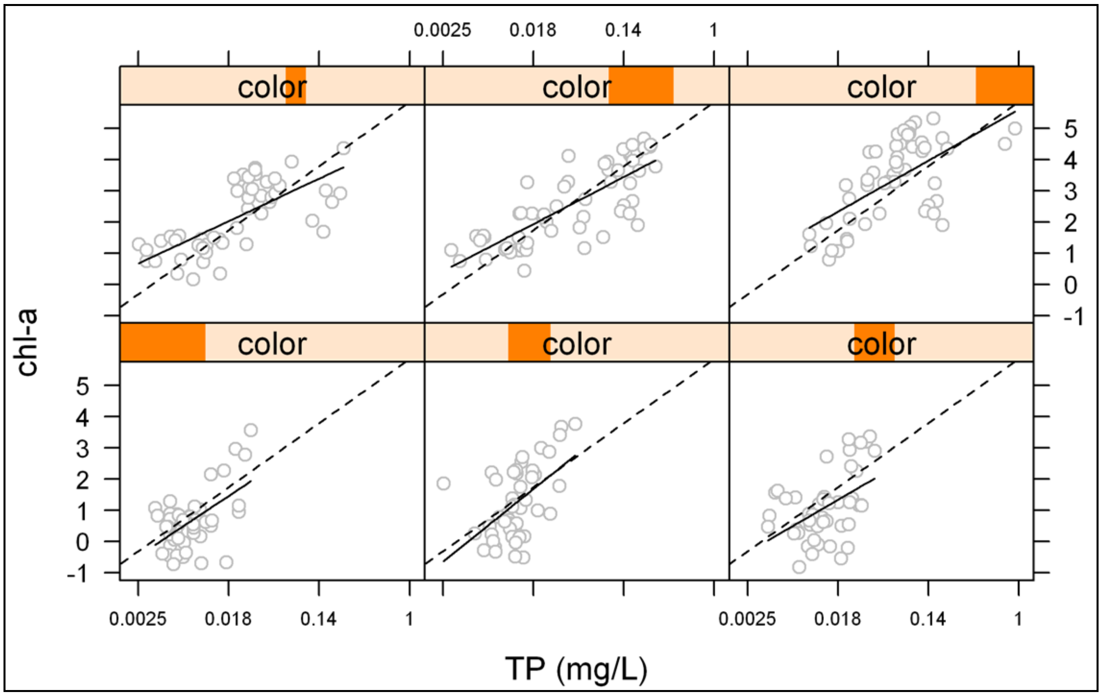

Figure 2-27. Relationships between TP and chl-*a* in different color subclasses of clear lakes. Note that the color associated with each subclass is shown as the orange bar, ranging from 6 to 40 PCU; the dashed line shows overall clear lake regression relationship and solid lines show relationships estimated within each class.

2.4.4 Calculation of Numeric Nutrient Criteria

Regression models describe the relationship between two variables where the magnitude of one variable (dependent) is a function of one or more independent variables. That is, a degree of the variance in the dependent variable is explained by the independent variable(s). The regression line and equation define the average relationship (i.e., for a given TN value in Figure 2-28, the average chl-*a* of all lakes with that concentration of TN is expected to fall on the regression line). The spread of points about the regression line shows the predictive uncertainty in the relationship for particular lakes or particular years—for a given TN value, some lakes or years will have a higher chl-*a* concentrations than the predicted value (regression line), and some will have a lower concentrations. This predictive uncertainty originates from differences in nutrient-chl-*a* relationships across different lakes and from variability in this relationship between years within the same lake.

Regression prediction intervals quantify the probability of different chl-*a* concentrations in individual lakes at specified concentrations of TN or TP. These prediction intervals incorporate the unexplained variability of chl-*a* as well as the uncertainty in the model parameters (slope and intercept). For nutrient criteria, predictive uncertainty can be managed by basing criteria on the points on different prediction intervals at which chl-*a* concentrations are equivalent to specified criteria (e.g., chl-*a* = 20 µg/L for colored lakes). For example, the 75th prediction interval (shown as the upper dashed line in Figure 2-28) can be interpreted as follows: at any given TN concentration, the model predicts that 75% of chl-*a* concentrations will be less than the 75th prediction interval. That is, the 75th prediction interval corresponds with a 25% probability of

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exceedance. Thus, to be protective and prevent exceedances of the chl-*a* criterion, the numeric nutrient criterion is set at a concentrations corresponding to the point at which the 75th prediction interval value is equivalent to the chl-*a* criterion.

Following this approach, EPA has defined the nutrient concentration yielding 25% probability of exceeding a chl-*a* target as the baseline criteria (A in Figure 2-28). Nutrient concentrations less than or equal to A (upper prediction interval) are unlikely to exceed the response threshold and therefore can be used as the basis for protective criteria. If there is no information on the biological condition of a lake (i.e., chl-*a* concentration), then the baseline criterion is the applicable criterion.

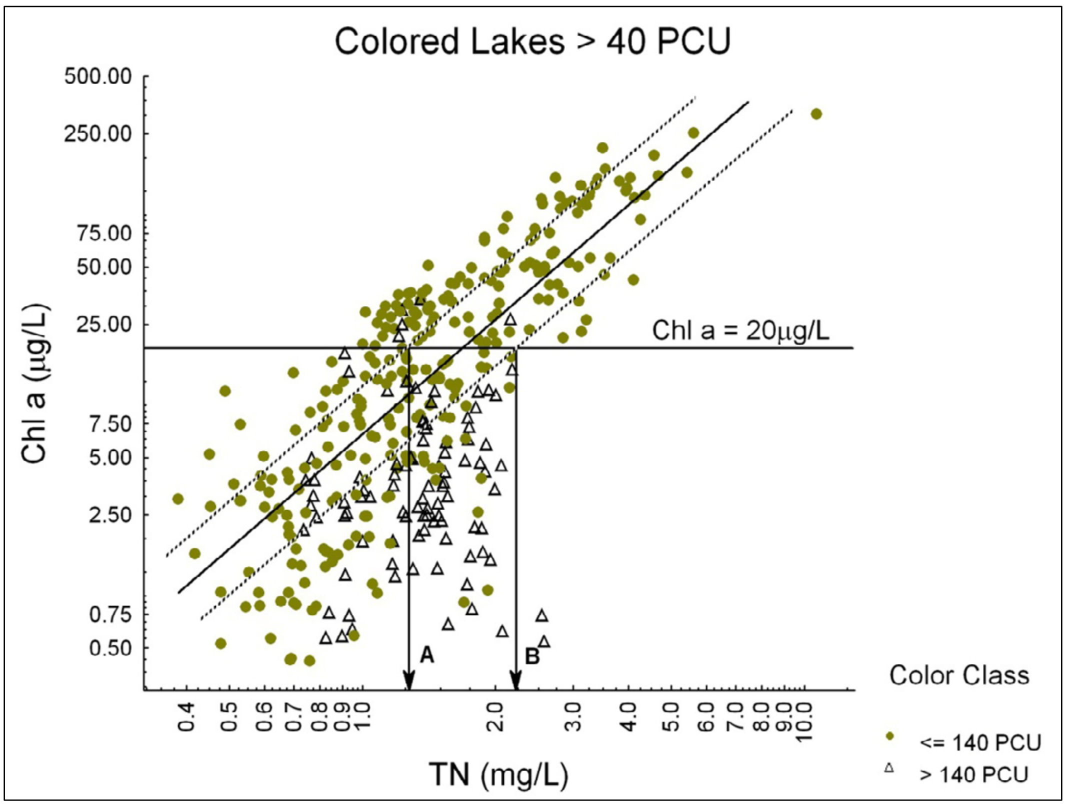

Figure 2-28. Chl-*a* and TN in colored lakes, showing 50% prediction interval (dotted lines) on each side of regression line.
Note that symbols and data as in Figure 2-23. Horizontal line: chl-*a* criterion (20 µg/L). A indicates TN concentration corresponding to 25% probability of chl-*a* exceeding 20 µg/L; B indicates concentration corresponding to 75% probability of exceedance.

This criteria framework provides flexibility for FDEP to derive lake-specific, modified TN, TP, and chl-*a* criteria provided the chl-*a* criteria are met. In such a case, the criteria for TN and/or TP may be modified within the specified range, in which the lower bound of the range is the aforementioned baseline criteria.

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Medium and high color lakes

Figure 2-28 and Figure 2-29 show all colored lakes, but the regression and criteria shown were developed from the intermediate colored lakes—between 40 and 140 PCU. Figure 2-28 and Figure 2-29 show the relationship between nutrients and chl-*a* in the highly colored lakes (> 140 PCU, open triangles). Examination of Figure 2-28 and Figure 2-29 indicates that more than 90% of highly colored lakes have average TN and TP concentrations below the modified criteria limit ("B" arrows), while 90% of these lakes also have chl-*a* concentrations less than 20 µg/L. Because most of these lakes appear to meet the modified criteria for intermediate colored lakes, there was no reason to develop and apply separate nutrient criteria for the highly colored lakes. Accordingly, the criteria table (Table 2-5) lists all colored lakes > 40 PCU.

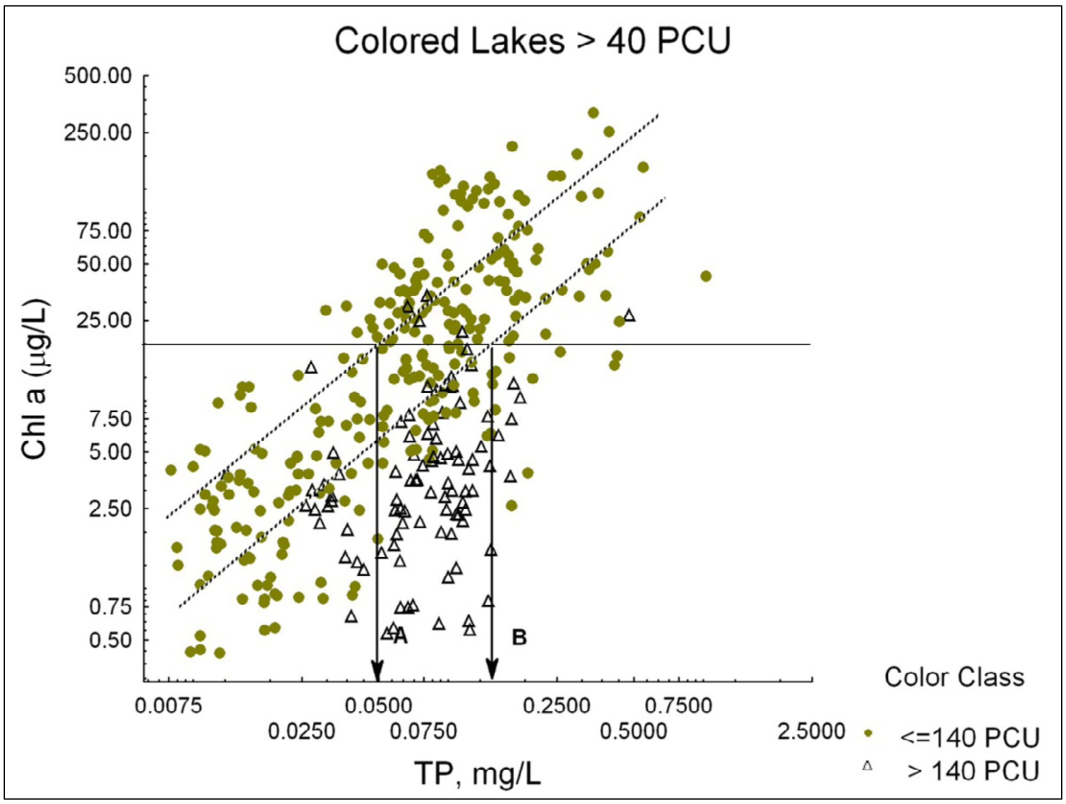

Figure 2-29. Chl-*a* and TP in colored lakes, showing 50% prediction interval (dotted lines). Note that symbols and data as in Figure 2-23. Horizontal line: chl-*a* criterion (20 µg/L). A indicates TN concentration corresponding to 25% probability of chl *a* exceeding 20 µg/L; B indicates concentration corresponding to 75% probability of exceedance.

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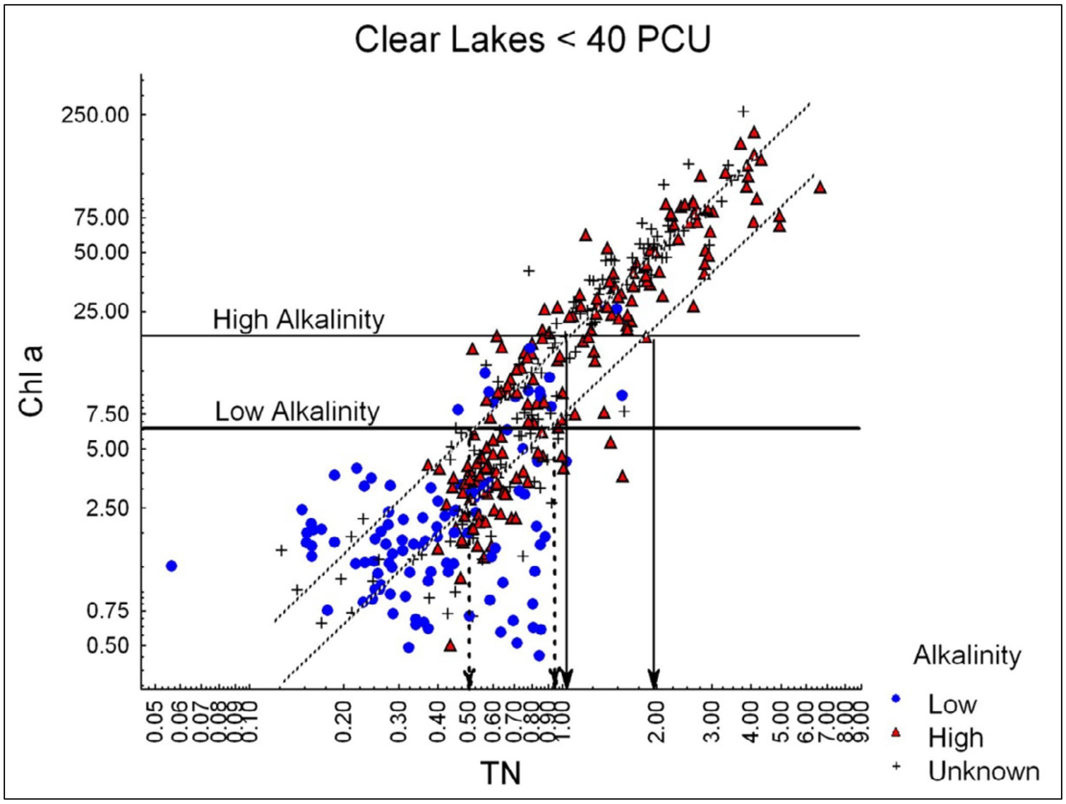

Figure 2-30. Chl-*a* and TN in clear lakes, showing 50% prediction interval (dotted lines). Note that symbols and data as in Figure 2-21. High alkalinity: chl-*a* criterion (20 µg/L) for clear, high alkalinity lakes; low alkalinity: chl-*a* criterion (6 µg/L) for clear, low alkalinity lakes. Solid arrows: criteria range for high alkalinity lakes; dashed arrows: criteria range for clear, low alkalinity lakes.

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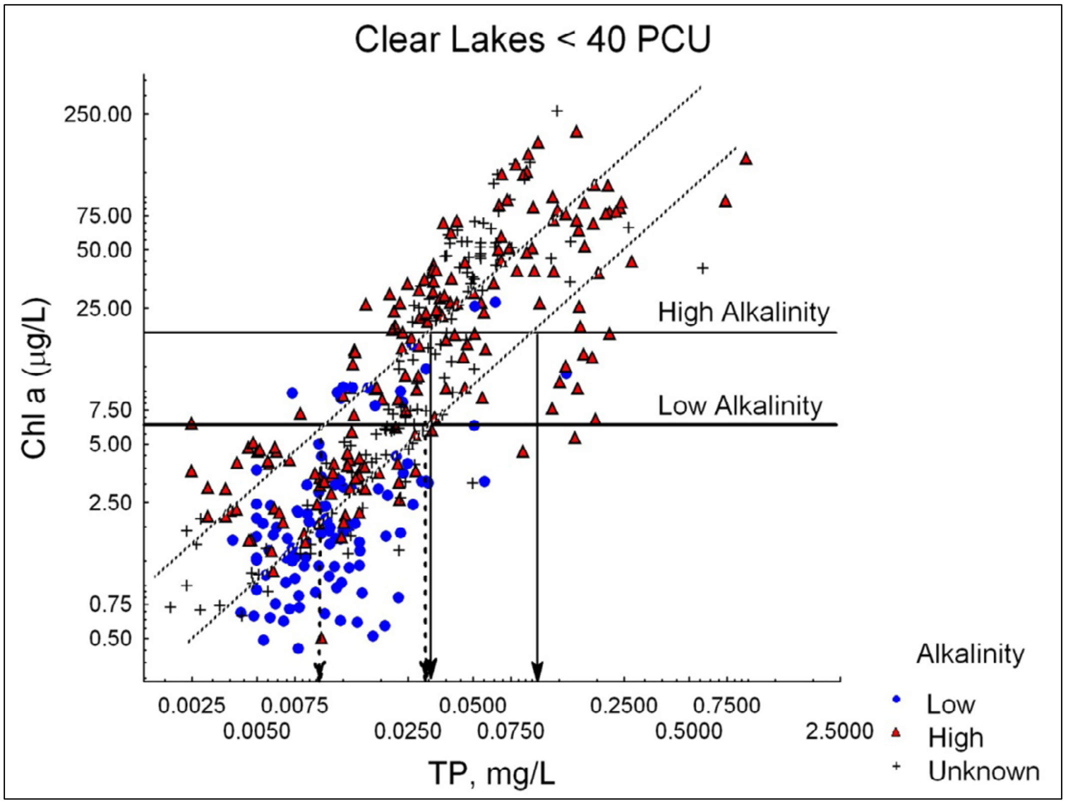

Figure 2-31. Chl-*a* and TP in clear lakes, showing 50% prediction interval (dotted lines). Note that symbols and data as in Figure 2-21. High alkalinity: chl-*a* criterion (20 µg/L) for high alkalinity lakes; low alkalinity: chl-*a* criterion (6 µg/L) for clear, low alkalinity lakes. Solid arrows: criteria range for high alkalinity lakes; dashed arrows: criteria range for clear, low alkalinity lakes.

2.4.5 Application of Lake-Specific Ambient Calculation Provision for Modified TN and TP Criteria

Because algal response is influenced by factors other than nutrients (grazing, macrophyte nutrient uptake, water retention time), there is uncertainty and variability in the response of any given lake to a particular nutrient concentration. Thus, it is reasonable to allow for the management for nutrients within the range of uncertainty. If data demonstrate that a lake is biologically healthy and does not experience excess algal growth (e.g., < 20 µg/L in a colored or clear, high alkalinity lake) despite having nutrient concentrations within the range of uncertainty, then the existing nutrient concentrations would appear to be reasonable.

Given the above approach—and using annual average chl-*a* values of 20 µg/L for colored and clear, high alkalinity lakes, and 6 µg/L for clear, low alkalinity Florida lakes, respectively—criteria ranges associated with protection of designated uses can be defined on the basis of the 50% prediction intervals (corresponding to 25% and 75% risk of exceeding chl a criteria) depicted in Figure 2-28 to Figure 2-31.

EPA has finalized a framework that uses both the upper and lower bounds of the 50% prediction interval to allow the derivation of modified, lake-specific TP and TN criteria to account for the natural variability in the relationship between chl-*a* response to TP and TN (Figure 2-28 to

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Figure 2-31). The framework promulgated in the final rule allows FDEP to calculate ambient modified criteria for TN and TP for a lake if the chl-*a* criterion is met in the three or more years of available ambient monitoring. As noted above, such alternative criteria must be based on at least three years of ambient monitoring data with (1) at least four measurements per year and (2) at least one measurement between May and September and one measurement between October and April each year. If a calculated modified TN and TP criterion is above the upper bound (upper 50% prediction interval), the upper bound is the criteria. Calculated modified TP and TN values may not exceed criteria applicable to streams to which a lake discharges.

The 50% prediction intervals and their derivation in the chl-*a* regression models are described above. The 50% prediction interval is the range within which one-half of chl-*a* observations are expected to fall for a given nutrient concentration (TN or TP), centered on the mean expectation at the regression line. In other words, the lower and upper bounds approximate the 25th and 75th percentiles of expected chl-*a* response for the given TN or TP, as predicted by the regression equations (Figure 2-28 through Figure 2-31).

One technical concern is the extent to which the variability in the data relating chl-*a* levels to TN and TP levels truly reflects differences among lakes, as opposed to temporal differences in the conditions in the same lake. EPA analyzed individual lake data within the entire lake data set to explore models of individual lake responses to nutrient enrichment. Models that incorporated individual lake response (single-lake models and Hierarchical Bayesian models that incorporate many lakes into the same model) verified that a large fraction of the total variability was due to "among lake" variability, and a small fraction was due to "within lake" variability.

Another technical concern is that a time lag might exist between the presence of high nutrients and the biological response. Lag time for changes of nutrient concentrations following change in loading is dependent on retention time. For example, Coveney et al. (2005) observed approximately twice the retention time. Biological responses follow the nutrient lag time. Because of the potential lag time, EPA based the range of TN and TP criteria on three consecutive annual geometric mean concentrations so the time lag in response would not be expected to affect the ambient assessment.

A third technical concern is the presence of temporary or long-term, site-specific factors that could suppress biological response. Examples include the presence of grazing zooplankton, excess sedimentation that blocks light penetration, extensive canopy cover, or seasonal herbicide use that impedes proliferation of algae. If any of those suppressing factors are removed, nutrient levels could result in a spike in algal production above protective levels. Again, the three-year data requirement will help minimize the effects of temporary site-specific factors.

EPA is requiring in its final rule that baseline criteria apply unless FDEP derives a modified criterion based on readily available and existing data and FDEP issues a determination that this modified criterion is applicable. The baseline criteria and the range within which baseline criteria can be modified are shown in Table 2-9. Criteria may be modified within the specified range if the following two conditions are satisfied. First, ambient chl-*a*, TN, and TP data are available, subject to the following data requirements: data for chl-*a*, TN, and TP are available from the immediately preceding three years (at a minimum) with (a) at least four measurements per year and (b) at least one measurement between May and September and one between October and

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April each year. Second, chl-*a* satisfies the criteria for the appropriate lake class (based on color and alkalinity). Modified criteria are then calculated using TN and TP data collected during the immediately preceding three year period. The values listed as the specified range are the lowest and the highest modified criterion that may be specified. Once established, modified criteria remain in place as the applicable water quality standards for all CWA purposes unless FDEP submits revisions to their standards.

Table 2-9. EPA's numeric nutrient criteria for Florida lakes.

Lake Color[a] and Alkalinity	Chl-*a* (mg/L)[b, *]	TN (mg/L)	TP (mg/L)
Colored Lakes[c]	0.020	1.27 [1.27–2.23]	0.05 [0.05–0.16]
Clear Lakes, High Alkalinity[d]	0.020	1.05 [1.05–1.91]	0.03 [0.03–0.09]
Clear Lakes, Low Alkalinity[e]	0.006	0.51 [0.51–0.93]	0.01 [0.01–0.03]

[a] Platinum Cobalt Units (PCU) assessed as true color free from turbidity.

[b] Chl-*a* is defined as corrected chlorophyll, or the concentration of chl-*a* remaining after the chlorophyll degradation product, phaeophytin *a*, has been subtracted from the uncorrected chl-*a* measurement.

[c] Long-term Color > 40 Platinum Cobalt Units (PCU)

[d] Long-term Color ≤ 40 PCU and Alkalinity > 20 mg/L $CaCO_3$

[e] Long-term Color ≤ 40 PCU and Alkalinity ≤ 20 mg/L $CaCO_3$

[*] For a given waterbody, the annual geometric mean of chl-*a*, TN or TP concentrations shall not exceed the applicable criterion concentration more than once in a three-year period.

EPA selected three years as a reasonable minimum length of time to appropriately account for anomalous conditions in any year that could lead to erroneous conclusions regarding the true relationship between nutrient levels in a lake and chl-*a* levels. EPA anticipates that the State would use at least three previous, consecutive years of data on record (i.e., it would not be appropriate to select three random years within a complete record). Data covering additional years, provided they meet FDEP's data quality objectives, would be appropriate, but data for all parameters would have to be represented. Such requirements would minimize the influence of long-term, site-specific factors and help ensure longer-term, stable conditions. Incorporating seasonal representation is important because nutrient levels can vary by season, and so an accurate estimate of annual average requires measurements in all seasons. In addition, this minimum sample size is conducive to deriving central tendency measurements, such as a geometric mean. The chl-*a* criterion must be met over the three or more years of ambient monitoring that define the record of observation for the lake to be eligible for the ambient calculation modified provision for TN and TP.

2.5 Duration and Frequency for Lake Criteria

Numeric criteria include magnitude (quantity), duration, and frequency components. For the chl-*a*, TN, and TP criteria for lakes, the criterion-magnitudes are expressed as annual geometric means, which includes duration (one-year averaging period). The criterion-frequency, or allowable excursion, is no more than once in a three year period.

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Appropriate duration and frequency components of criteria should be based on how the data used to derive the criteria were analyzed and what the implications are for protecting designated uses given the effects of exposure at the specified criterion concentration for different periods and recurrence patterns. For lakes, the stressor-response relationship was based on an annual geometric mean of individual years at individual lakes. The appropriate period is therefore annual. The key question is whether this annual geometric mean needs to be met every year or if some allowance for a particular year to exceed the applicable criterion could still be considered protective.

Nutrient criteria, unlike criteria for toxic pollutants in most cases, are typically established within the range of natural variability. A temporary spike in nutrient levels does not necessarily harm the aquatic resource. In fact, natural systems have evolved to process variable inputs of nutrients, particularly where there is much natural variability in hydrologic conditions and precipitation patterns (e.g., wet years, dry years). Although biological response in the form of algal production, measured by chl-*a*, can appear very quickly, longer term shifts in biological conditions, such as loss of underwater grasses, do not occur as the result of a single event or conditions in a single year. If grasses experience reduced light during a portion of a year, the effect on them is not expected to be as large or persistent as if reduced light conditions occur in multiple years. In other words, severe nutrient impacts often result from chronic exposure to elevated nutrients. In addition, extreme incidents of nutrient pollution are not likely to be isolated incidents. Severe exposure to high nutrients is most likely in the context of a waterbody that has chronic nutrient pollution.

EPA considered studies by Walker (1984) and Bachmann et al. (2003), which examined bloom frequency with more intensive data sets. Their analyses predicted that an annual mean of 20 µg/L chl-*a* could result in blooms greater than 30 µg/L chl-*a* for 10 to 18% of the time. EPA also considered the work by Long et al. (2009), which suggested water transparency as a critical value to protect in the MEI model development. For the current rulemaking, data were not sufficient to extend either of these considerations to the entire State and all lake types. However, EPA believes they may be useful for future refinements of nutrient criteria such as the development of lake-specific numeric nutrient criteria.

Data that contributed to the analysis of response to nutrients, as well as data generated from supporting paleolimnological studies and MEI modeling, typically represent periods greater than a single year. Conceptually, trophic state and the categories of oligotrophic, mesotrophic, and eutrophic, are attributes that are long-term (> 1 year) and change with the long-term loading rates to a system. As an example, the eutrophication of Lake Apopka and subsequent nutrient management and ongoing recovery of the lake are processes that are taking years to decades (Coveney et al. 2005). Moreover, many of the models and analyses that form the basis of the results above were designed to represent the "steady state," or long-term stable water quality conditions. Thus, natural variability has already been factored into the calculated criteria for lakes. They reflect central tendencies (with associated variability) of data, with decisions made about the distribution and variability surrounding those central tendency values.

Some researchers have suggested caution in applying steady-state assumptions to lakes with long residence times (Kenney 1990). That is, the effects of spikes in annual loading could linger and disrupt the steady state in some lakes. As a result, and as noted above, EPA is finalizing an

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excursion frequency for the annual geometric mean for lakes not to exceed more than once in three years.

Those duration and frequency components take into account that hydrological variability will in turn produce variability in measured nutrient concentrations and that individual measurements might exceed the criteria. Furthermore, these components balance the representation of underlying data and analyses on the basis of central tendency of many years of data (i.e., the long-term average component) with the need to exercise some caution to ensure that lakes have sufficient time to process individual years of elevated nutrient levels and avoid the possibility of cumulative and chronic effects (i.e., no more than once in a 3-years).

2.6 Downstream Protection of Lakes

An important component of setting appropriately protective water quality criteria is ensuring that the criteria also provide for the attainment and maintenance of the water quality standards of downstream waters (40 CFR 131.10(b)). Therefore, the lake criteria described above affect the derivation of stream criteria described in Chapter 1. This section describes EPA's approach in considering the downstream water quality criteria EPA derived to protect Florida's lakes. EPA's analyses for deriving numeric nutrient criteria for streams in Florida, as reflected in the final rule, indicate that the final criteria values for instream protection of streams may not be sufficient to fully protect downstream lakes (e.g., stream TP for the West Central NWR > lake TP for all three lake classes). EPA's criteria for lakes can be, in some cases, more stringent than the final criteria for streams that flow into the lakes. The applicable downstream protective value (DPV) applies to streams at the point of entry into the lake. If the DPV is not attained at the point of entry into the lake, then the collective set of streams network in the upstream watershed does not attain the DPV, which is an applicable water quality criterion for the water segments in the upstream watershed. The State or EPA may establish additional DPVs at upstream tributary locations that are consistent with attaining the DPV at the point of entry into the lake. The State or EPA also have discretion to establish DPVs to account for a larger watershed area (i.e., include waters beyond the point of reaching waterbodies that are not streams as defined by this rule).

2.6.1 Approach

EPA is providing for a flexible downstream protection approach for lakes that contains several alternative approaches to ensure that the applicable criteria for streams that flow into lakes provide for the attainment and maintenance of the applicable lake criteria. First, EPA is specifying that, where sufficient data and information is available, the U.S. Army Corps of Engineers BATHTUB reservoir model (Walker 1999) may be used to compute downstream protective values (DPV) necessary in the stream to protect a given lake. Second, EPA is allowing the use of scientifically defensible models other than BATHTUB, such as the Water Quality Analysis Simulation Program (WASP, see Section 2.6.3), that demonstrate protection of water quality standards in downstream lakes. Third, in the absence of sufficient data to calibrate BATHTUB or other models, EPA is specifying that default TN and TP criteria shall be applied to streams that flow into the lake. The first default provides that, where the downstream lake attains its applicable chl-*a* and TP and/or TN criteria, then the DPV for TN and/or TP would be is the ambient instream level of TN and/or TP at the point of entry into the lake associated with measurements that demonstrate attainment of applicable TN and TP criteria in the lake, down to

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but not more stringent than the TN and/or TP criteria for the lake. Degradation in water quality from this ambient condition DPV will be considered non-attainment of the DPV, unless the DPV is adjusted use of a scientifically defensible model such as BATHTUB. The second default provides that where the downstream lake either does not attain the applicable TN, TP, and/or chl-*a* criteria or has not been assessed, then the DPV for TN and/or TP is the applicable TN and TP criteria for the downstream lake.

For all of these approaches, the final applicable criteria for the stream at the point of entry into the lake would be the more stringent of either the instream criteria or the DPVs derived using one of the three approaches.

EPA believes BATHTUB is appropriate for simplified DPV calculations because BATHTUB can represent a number of site-specific variables that may influence nutrient responses. In addition, BATHTUB can estimate TN concentrations. The BATHTUB model has been previously used for lake water quality management purposes—both nationally and in the State of Florida. That model was selected because, while it does not have extensive data requirements, it does provide for the capability to be calibrated based on observed site-specific lake data and it provides for reliable estimates that will ensure the protection of downstream lakes.

EPA also specifically authorizes FDEP or EPA to use a model other than BATHTUB when either determines that it would be appropriate to use another scientifically defensible technical model that demonstrates protection of downstream lakes. While BATHTUB is a peer-reviewed and versatile model, there are other models that, when appropriately calibrated and applied, can offer additional capability to address more complex situations and address an even greater degree of site-specificity.

One example of an alternative model that FDEP or EPA might consider using for particularly complex site-specific conditions is the WASP model (Wool et al. 2001). That model allows users to conduct detailed simulations of water quality responses to natural and manmade pollutant inputs. The WASP model is a dynamic (time-varying) compartment-modeling program for aquatic systems, including both the water column and the underlying benthos. The model allows the user to simulate systems in 1, 2, or 3 dimensions and with a variety of pollutant types. The model can represent time varying processes of advection, dispersion, point and diffuse mass loading, and boundary exchange. It also can be linked with hydrodynamic and sediment transport models that can provide flows, depths, velocities, temperature, salinity and sediment fluxes. Additional technical information may be found online at http://www.epa.gov/athens/wwqtsc/html/wasp.html. Like BATHTUB, WASP has also been previously used for lake water quality management purposes, both nationally and in the State of Florida.

2.6.2 BATHTUB

The BATHTUB model is designed to apply empirical nutrient mass balance and eutrophication relationships to both simple and morphometrically complex lakes and reservoirs. The program performs steady-state water and nutrient mass balance calculations, uses spatially segmented hydraulic networks, and accounts for advective and diffusive transport of nutrients. When properly calibrated and applied, BATHTUB predicts annual or seasonally averaged nutrient-

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related water quality conditions such as TP, TN, and chl-*a* concentrations, transparency, and hypolimnetic oxygen depletion rates. The model can be applied to a variety of lake sizes, shapes, and transport characteristics. A high degree of flexibility is available for specifying model segments as well as multiple influent streams. Because water quality conditions are calculated from influent loadings using empirically-derived relationships, BATHTUB implicitly accounts for internal recycling of nutrients from bottom sediments. Additional technical references are available that describe the model, its development, and its applications (Baniukiewicz and Gilbert 2004; Gao 2006; Gao and Gilbert 2003; Gao and Gilbert 2006; James et al. 2007; Kennedy 1995; Robertson and Rose 2008; Shelley et al. 2005; Walker 1981, 1982, 1999). The BATHTUB has also been studied elsewhere in the scientific literature (James et al. 2002; Mankin et al. 2003; Muhammetoglu et al. 2005; Smeltzer and Quinn 1996; Yoko et al. 1997).

To reflect data limitations or other sources of uncertainty, key inputs to the model can be specified in probabilistic terms (mean and coefficient of variation, CV). Outputs are expressed in terms of a mean value and CV for each mass balance term and response variable. Output CVs are based upon a first-order error analysis that accounts for input variable uncertainty and inherent model error.

Usage/calibration

The BATHTUB model is available online from the U.S. Army Corps of Engineers Waterways Experiment Station at http://el.erdc.usace.army.mil/products.cfm?Topic=model&Type=watqual. Application of the BATHTUB model to calculate DPVs consists of two steps, (1) development of a calibrated representation of lake response based on observed loads of water and nutrients; and (2) adjustment of influent concentrations until the specified in-lake criteria for TN, TP, and chl-*a* are predicted to be attained.

Calibration of BATHTUB can take place at varying levels of complexity depending upon the availability of data. The user must first assemble or estimate available information on lake morphometry, inflow, outflow, and observed water quality. For many smaller Florida lakes, representation of the waterbody as a single mixed segment may be sufficient; however, multiple segment models can be developed for more hydraulically complex lakes.

The general sequence for the calibration of BATHTUB is described by Walker (1987, 1999) and includes the following steps:

1. Determine appropriate averaging period over which water and mass balance calculations are performed. This may be annual or seasonal. Criteria in terms of the calculated nutrient turnover ratio are provided by Walker.
2. Calibrate mass balance for water.
3. (Optional) For multi-segment lake representations, calibrate rates of constituent exchange between lake segments for a conservative tracer.
4. Calibrate mass balance for nutrients over the appropriate annual or seasonal averaging period. Calibration factors should be adjusted only within the ranges recommended by Walker.

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5. After obtaining a satisfactory representation of growing season nutrient concentrations, calibrate chl-*a* concentrations and Secchi depth.

Where sufficient data are available, application to multiple years is recommended.

Following development of a satisfactory model, the influent concentrations are varied to achieve the in lake criteria for TN, TP, and chl-*a* (as a function of TN and TP). The most restrictive concentration values are then selected as DPVs. The BATHTUB model has been widely applied for the development of nutrient TMDLs in Florida and is a tested and accepted approach suitable to the estimation of site-specific DPVs.

2.6.3 WASP

EPA also examined use of the WASP model (Wool et al. 2001) for establishing DPVs. It is a sophisticated temporally dynamic, mechanistic model that can be used to predict time series of nutrients and chl-*a* within a lake in response to time series inputs. WASP allows the individual characteristics of a lake to be represented in the simulation (retention time, light, color, pH, nutrient concentrations, and interaction with benthos). The model can also be run in a "steady-state" condition representing a critical period of time (high retention time, summer season), or in a time-variable condition where long-term average and exceedance frequencies can be examined. The WASP has also been applied and studied elsewhere in the scientific literature (James et al. 1997; James et al. 2005; Kiesling et al. 2001; Pauer et al. 2007; Petrus 2007; Reckhow 1999; Stansbury and Admiraal 2004; Steinman and Rosen 2000; Wang et al. 2009).

Application of the WASP model generally requires detailed site-specific information and a significant calibration effort. Time series output (e.g., daily, weekly) must be converted to growing season averages for comparison to nutrient criteria, and iteration to determine influent concentrations that achieve instream criteria can be complex and laborious. A full implementation of WASP for site-specific use typically involves multiple segments and multiple calibration endpoints, with run times for a 10-year simulation on the order of 30 minutes to an hour.

The WASP model is an appropriate alternative to BATHTUB for setting DPVs, particularly where complex site-specific conditions are not adequately addressed by BATHTUB.

Application

Application of WASP requires calibration. Calibration can be time-consuming, in part because the model has a large number of highly interrelated parameters. The effort per lake in applying WASP is relatively large. A better approach could be to apply a simpler tool (such as BATHTUB) to obtain an initial estimate of DPVs, then modify these, where necessary, on a site-specific basis using a fully developed and rigorously calibrated lake or reservoir model.

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2.7 Summary

2.7.1 Lake Classification

The natural diversity of Florida's lakes requires a classification system so that appropriate numeric nutrient criteria can be developed. Florida's geology, topography, and climate define the conditions in which the state's lakes formed, and influence their natural conditions. Florida is close to sea-level and has generally flat topography, a wet climate, and is underlain by limestone throughout much of the State. Many of Florida's lakes are solution lakes—the result of dissolution and collapse of caverns in the underlying rock, forming sinkholes. The wet climate and low topography ensure that the water table is near the surface. Many of Florida's lakes are shallow. The limestone bedrock may occur at the surface, and in some areas springs are present, especially in areas where the Floridan Aquifer is exposed at the surface. In other regions, the limestone may be overlain by other geologic formations such as marine sands and clays that prevent the water in lakes from interacting with the limestone bedrock. Limestone is often associated with phosphorus, and in Florida the widespread Hawthorn Group is a source of phosphate rock that is an economic resource.

Florida lakes range widely in natural alkalinity, from truly acidic lakes (pH<5) with almost no measurable alkalinity, to those that occur in areas with carbonate-rich (limestone) sediments. Florida also has extensive wetlands, and owing to hydrologic connection to, or isolation from, the wetlands, lakes may range from those with clear water to waters that are heavily colored. The water color is due to dissolved organic matter (e.g., tannins, lignins, and their decomposition products) from decaying wetland vegetation. Water color affects the penetration of light in the water, and is also associated with organic nitrogen, which contributes to TN.

EPA's classification of Florida's lakes in this document is supported previous classifications, and revealed the importance of color and alkalinity in identifying lake type and for predicting the response of chl-*a* to N and P enrichment. EPA's classification identified the following four lake types: clear, low alkalinity lakes; clear, high alkalinity lakes; highly-colored lakes; and intermediate-colored lakes. The numeric nutrient criteria for the highly-colored and intermediate-colored lakes are the same.

2.7.2 Trophic State and Aquatic Life Use Protection

EPA is using chl-*a* concentration as an indicator of a healthy biological condition, supportive of natural balanced populations of aquatic flora and fauna in each of the classes of Florida's lakes. Excess algal growth is associated with degradation in aquatic life and chl-*a* levels are a measure of algal growth. EPA used trophic status, existing distributions of chl-*a* concentrations in all sampled lakes, and chl-*a* concentrations observed in least-disturbed lakes as the primary lines of evidence to derive chl-*a* concentrations that would be protective of natural balanced populations of aquatic flora and fauna in lakes.

Lakes can be classified into one of three trophic state categories (i.e., oligotrophic, mesotrophic, and eutrophic). After considering previously published data analyses of subtropical and tropical lakes, and Florida lakes, and analyzing data collected from Florida lakes, EPA concluded that mesotrophic status is the appropriate expectation for colored lakes, as well as clear, high alkalinity lakes. This is because these lakes receive significant natural nutrient input and still support a healthy diversity of aquatic life in warm, productive climates such as Florida. However, clear, low alkalinity lakes in Florida do not receive comparable natural nutrient input; thus, EPA concluded that oligotrophic status is the appropriate expectation in those lakes to support balanced natural populations of aquatic flora and fauna.

Existing distributions of chl-*a* concentrations in all sampled lakes and distributions of chl-*a* concentrations in least-disturbed lakes also informed criteria derivation. Differences in the number of least-disturbed lakes available in each of the classes led to differences in how this information was used to derive criteria, but in all cases, least-disturbed lakes provided a context for EPA to interpret existing distributions of chl-*a* concentrations in all sampled lakes.

2.7.3 Numeric TP and TN Criteria Development

EPA examined predictive relationships between nutrients and the chl-*a* response. Given the lake classification, and considerations for implementation of criteria, EPA derived criteria from univariate simple linear regressions of chl-*a* with TN and TP. The upper 50% prediction intervals around the regression lines defined a 25% probability of exceeding the given chl-*a* criteria (6 or 20 µg/L), which was taken as the baseline criteria. The lower prediction interval was taken as the upper nutrient limit for modified criteria (see criteria explanation, Table 2-9).

2.7.4 Duration and Frequency

Aquatic life water quality criteria include magnitude, duration, and frequency components. EPA is finalizing the criterion-frequency as no more than once-in a three-year excursion frequency of the annual geometric mean criteria for lakes. The duration and frequency components of the criteria are consistent with the data set used to derive these criteria, which applied stressor-response relationships based on annual geometric means for individual years at individual lakes.

These selected duration and frequency components recognize that hydrological variability (e.g., high and low flows) will produce variability in nutrient concentrations, and that individual measurements may at times be greater than the criteria magnitude concentrations without causing unacceptable effects to aquatic organisms and their uses. Furthermore, these duration and frequency components balance the representation of underlying data and analyses based on the central tendency of many years of data with the need to exercise some caution to ensure that lakes have sufficient time to process individual years of elevated nutrient levels and avoid the possibility of cumulative and chronic effects (i.e., no more than one excursion component)

2.7.5 Downstream Protection of Lakes

An important component of setting appropriately protective water quality criteria is ensuring that the criteria also provide for the attainment and maintenance of the water quality standards of downstream waters (40 CFR 131.10(b)). This position is supported by the scientific literature, which has concluded that the source of many adverse water quality impacts can be traced to degradation of water quality upstream of the impacted aquatic system (e.g., U.S. EPA 2007; Paerl, 2009; Conley et al. 2009). EPA's analyses for deriving numeric nutrient criteria for streams in Florida indicate that the final criteria values for instream protection of streams may not be sufficient to fully protect downstream lakes. Thus, EPA's criteria for lakes are, in some cases, more stringent than the final criteria for streams that flow into the lakes.

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EPA is providing for a flexible downstream protection approach for lakes that contains several alternative approaches to ensure that the applicable criteria for streams that flow into lakes provides for the attainment and maintenance of the applicable lake criteria. First, EPA is specifying that the U.S. Army Corps of Engineers BATHTUB reservoir model (Walker, 1999) may be used to compute downstream protective values (DPV) necessary in the stream to protect a given lake. Second, EPA is allowing the use of scientifically defensible models other than BATHTUB that demonstrate protection of WQS in downstream lakes. Third, in the absence of sufficient data to calibrate BATHTUB or other models, EPA is specifying how to derive default DPV for TN and TP. DPVs must be attained at the point where the stream empties into the downstream lake. For all of these approaches, the final applicable criteria for the stream that flows into the lake would be the more stringent of either the instream criteria as described in Chapter 1 or the DPVs derived using one of the three approaches.

2.8 References

ADEQ (Arizona Department of Environmental Quality). 2008. *Notice of Final Rulemaking*. *Title 18. Environmental Quality Chapter 11, Department of Environmental Quality Water Quality Standards*. Arizona Department of Environmental Quality, Phoenix, AZ. <http://www.azdeq.gov/function/laws/download/NFRM1.pdf>. Accessed January 2010.

Arce, R.G., and C.E. Boyd. 1975. Effects of agricultural limestone on water chemistry, phytoplankton productivity, and fish production in soft water ponds. *Transactions of the American Fisheries Society* 2:308–312.

Bachmann, R.W., D.L. Bigham, M.V. Hoyer, and D.E. Canfield, Jr. N.D. To what extent do natural versus anthropogenic factors determine the concentrations of total phosphorus, total nitrogen and chlorophyll in Florida lakes? Draft manuscript, submitted as part of comment letter 1895.1 to Docket ID No. EPA-HQ-OW-2009-0596.

Bachmann, R.W., M.V. Hoyer, and D.E. Canfield, Jr. 1999. The restoration of Lake Apopka in relation to alternative stable states. *Hydrobiologia* 394:219–232.

Bachmann, R.W., C.A. Horsburgh, M.V. Hoyer, L.K. Mataraza, and D.E. Canfield, Jr. 2002. Relations between trophic state indicators and plant biomass in Florida lakes. *Hydrobiologia* 470:219–234.

Bachmann, R.W., M.V. Hoyer, and D. E. Canfield, Jr. 2003. Predicting the frequencies of high chlorophyll levels in Florida lakes from average chlorophyll or nutrient data. *Lake and Reservoir Management* 19(3):229–241.

Baniukiewicz, A., and D. Gilbert. 2004. Nutrient TMDL for Lake Hunter. Florida Department of Environmental Protection, Southwest District. <http://www.dep.state.fl.us/water/tmdl>.

Barnese, L.E., and C.L. Schelske. 1994. Effects of nitrogen, phosphorus and carbon enrichment on planktonic and periphytic algae in a softwater, oligotrophic lake in Florida, USA. *Hydrobiologia* 277(3):159–170.

US EPA ARCHIVE DOCUMENT

US EPA ARCHIVE DOCUMENT

Brown, C.D., M.V. Hoyer, R.W. Bachmann, and D.E. Canfield, Jr. 2000. Nutrient–chlorophyll relationships: an evaluation of empirical nutrient–chlorophyll models using Florida and north-temperate lake data. *Canadian Journal of Fisheries and Aquatic Sciences* 57:1574–1583.

Canfield, D.E., Jr., M.J. Maceina, L.M. Hodgson, and K.A. Langeland. 1983. Limnological features of some northwestern Florida lakes. *Journal of Freshwater Ecology* 2(1):67–79.

Carlson, R.E. 1977. A trophic state index for lakes. *Limnology and Oceanography* 22:361–369.

Carlson, R.E., and J. Simpson. 1996. *A Coordinator's Guide to Volunteer Lake Monitoring Methods*. North American Lake Management Society, Madison, WI.

Conley, D.J., H.W. Paerl, R.W. Howarth, D. F. Boesch, S.P. Seitzinger, K.E. Havens, C. Lancelot, G.E. Likens. 2009. Controlling Eutrophication: Nitrogen and Phosphorus. *Science* 323:1014–1015.

Coveney, M.F., E.F. Lowe, L.E. Battoe, E.R. Marzolf, and R. Conrow. 2005. Response of a eutrophic, shallow subtropical lake to reduced nutrient loading. *Freshwater Biology* 50:1718–1730.

Downing, J.A., S.B. Watson, and E. McCauley. 2001. Predicting cyanobacteria dominance in lakes. *Canadian Journal of Fisheries and Aquatic Sciences* 58:1905–1908.

Edmondson, W.T. 1994. Sixty years of Lake Washington: a curriculum vitae. *Lake and Reservoir Management* 10(2):75–84.

Elser, J.J., E.R. Marzolf, and C.R. Goldman. 1990. Phosphorus and nitrogen limitation of phytoplankton growth in the freshwaters of North America: a review and critique of experimental enrichments. *Canadian Journal of Fisheries and Aquatic Science* 47:1468–1477.

Elser, J.J., M.E.S. Bracken, E.E. Cleland, D.S. Gruner, W.S. Harpole, H. Hillebrand, J.T. Ngai, E.W. Seabloom, J.B. Shurin, and J.E. Smith. 2007. Global analysis of nitrogen and phosphorus limitation of primary producers in freshwater, marine and terrestrial ecosystems. *Ecology Letters* 10: 1135–1142.

Elser, J.J, T. Andersen, J.S. Baron, A.-K. Bergström, M. Jansson, M. Kyle, K.R. Nydick, L. Steger, and D.O. Hessen. 2009. Shifts in lake N:P stoichiometry and nutrient limitation driven by atmospheric nitrogen deposition. *Science* 326:835–837.

Fairchild, G.W., and J.W. Sherman. 1993. Algal periphyton response to acidity and nutrients in softwater lakes: Lake comparison vs. nutrient enrichment. *Journal of the North American Benthological Society* 12(2):157–167

FDEP (Florida Department of Environmental Protection). 2009. *Technical Support Document: Development of Numeric Nutrient Criteria for Florida Lakes and Streams*. Florida Department of Environmental Protection, Tallahassee, FL. June 2009. <http://www.dep.state.fl.us/water/wqssp/nutrients/docs/tsd_nutrient_crit.docx>. Accessed September 6, 2010.

Fore, L. 2007. *An Evaluation of Benthic Macroinvertebrate Assemblages as Indicators of Lake Condition*. Final Report to the Florida Department of Environmental Protection, Tallahassee, FL.

Fore, L.S., R. Frydenborg, N. Wellendorf, J. Espy, T. Frick, D. Whiting, J. Jackson, and J. Patronis. 2007. *Assessing the Biological Condition of Florida's Lakes: Development of the Lake Vegetation Index (LVI)*. Report to the Florida Department of Environmental Protection, Tallahassee, FL. <ftp://ftp.dep.state.fl.us/pub/labs/assessment/sopdoc/lvi_final07.pdf>. Accessed January 2010.

Freund, R.J., W.J. Wilson, and P. Sa. 2006. *Regression Analysis: Statistical Modeling of a Response Variable*. 2nd Ed. Academic Press. Burlington, MA, USA. Pp 65:68.

Gao, X. 2006. Nutrient and Unionized Ammonia TMDLs for Lake Jesup, WBIDs 2981 and 2981A. Florida Department of Environmental Protection, Central District, Middle St. Johns Basin. < http://www.dep.state.fl.us/water/tmdl>.

Gao, X., and D. Gilbert. 2003. Nutrient Total Maximum Daily Load for Orange Lake, Alachua County, Florida. Watershed Assessment Section Florida Department of Environmental Protection 2600 Blair Stone Road, MS 3555 Tallahassee, FL 32399-2400 <http://www.dep.state.fl.us/water/tmdl >.

Gao, X., and D. Gilbert. 2006. Nutrient Total Maximum Daily Load For Trout Lake, Lake County, Florida. Watershed Assessment Section Florida Department of Environmental Protection 2600 Blair Stone Road, MS 3555 Tallahassee, FL 32399-2400. Geiger, N.S. 2000. Epiphytic algae on deep-dwelling bryophytes in Waldo Lake, Oregon. *Lake and Reservoir Management* 16:100–107.

Gerritsen, J., B. Jessup, E.W. Leppo, and J. White. 2000. *Development of Lake Condition Indexes (LCI) for Florida*. Prepared for the Florida Department of Environmental Protection by Tetra Tech, Inc., Owings Mills, MD.

Gill, J.L. 1987. Biases in regression when prediction is inverse to causation. *Journal of Animal Science* 64:594–600.

Gregory, J. 2007. *Lessons Learned in Nutrient Criteria Development in Virginia.* Virginia Department of Environmental Quality, Richmond, VA.

Griffith, G.E., D.E. Canfield Jr., C.A. Horsburgh, J.M. Omernik, and S.H. Azevedo. 1997. *Lake Regions of Florida*. U.S. Environmental Protection Agency, Corvallis, OR. <http://www.epa.gov/wed/pages/ecoregions/fl_eco.htm>. Accessed October 2009.

US EPA ARCHIVE DOCUMENT

Grubbs, G. 2001. U.S. EPA. (Memorandum to Directors of State Water Programs, Directors of Great Water Body Programs, Directors of Authorized Tribal Water Quality Standards Programs and State and Interstate Water Pollution Control Administrators on Development and Adoption of Nutrient Criteria into Water Quality Standards. November 14, 2001.

Grumbles, B.H. 2007. U.S. EPA. (Memorandum to Directors of State Water Programs, Directors of Great Water Body Programs, Directors of Authorized Tribal Water Quality Standards Programs and State and Interstate Water Pollution Control Administrators on Nutrient Pollution and Numeric Water Quality Standards. May 25, 2007.

Guildford, S.J., and R.E. Hecky. 2000. Total nitrogen, total phosphorus, and nutrient limitation in lakes and oceans: Is there a common relationship? *Limnology and Oceanography* 45:1213–1223.

Harrell, F.E. 2001 Regression Modeling Strategies: With Applications To Linear Models, Logistic Regression, And Survival Analysis. Springer Series in Statistics. Springer.

Havens, K. 2000. Using Trophic State Index (TSI) values to draw inferences regarding phytoplankton limiting factors and seston: composition from routine water quality monitoring data. *Korean Journal of Limnology* 33(3):187–196.

Havens, K.E., L.A. Bull, G.L. Warren, T.L. Crisman, E.J. Phlips, and J.P. Smith. 1996. Food web structure in a subtropical lake ecosystem. *Oikos* 75(1):20–32.

Hein, M. 1997. Inorganic carbon limitation of photosynthesis in lake phytoplankton. *Freshwater Biology* 37(3):545–552.

Heiskary, S., and B. Wilson. 2008. Minnesota's approach to lake nutrient criteria development. *Lake and Reservoir Management* 24:282–297.

Herlihy, A.T., J.L. Stoddard, and C.B. Johnson. 1998. The relationships between stream chemistry and watershed land use in the Mid-Atlantic Region. *U.S. Water, Air, and Soil Pollution* 105:377–386.

Hoyer, M.V., C.D. Brown, and D.E. Canfield, Jr. 2004. Relations between water chemistry and water quality as defined by lake users in Florida. *Lake and Reservoir Management* 20:240–248.

James, R.T., J. Martin, T. Wool, and P. F. Wang. 1997. A sediment resuspension and water quality model of Lake Okeechobee. *Journal of American Water Resources Association*. 33(3):661–680.

James, R.T., V.J. Bierman, M.J. Erickson, and S.C. Hinz. 2005. The Lake Okeechobee water quality model (LOWQM) enhancements, calibration, validation and analysis. *Lake and Reservoir Management* 21(3):231–260.

US EPA ARCHIVE DOCUMENT

James, W.F., J.W. Barko, H.L. Eakin, and P.W. Sorge. 2002. Phosphorus budget and management strategies for an urban Wisconsin lake. *Lake and Reservoir Management* 18(2):149–163.

James, W.F., A. Dechamps, N. Turyk, and P. McGinley. 2007. Contribution of *Potamogeton crispus* Decay to the Phosphorus Budget of McGinnis Lake, Wisconsin. U.S. Army Engineer Research and Development Center Geotechnical and Structures Laboratory 3909 Halls Ferry Road Vicksburg, MS 39180-6199. ERDC/TN APCRP-EA-15.

Johnson, A.C., and R.W. Castenholz. 2000. Preliminary observations of the benthic cyanobacteria of Waldo Lake and their potential contribution to lake productivity. *Lake and Reservoir Management* 16:85–90.

Kiesling, R.L., A.M.S. McFarland, and L.M. Hauck. 2001. Nutrient Targets for Lake Waco and North Bosque River: Developing Ecosystem Restoration Criteria. Texas Institute for Applied Environmental Research, Tarleton State University, Stephenville, Texas.

Kennedy, R.H. 1995. *Application of the BATHTUB Model to Selected Southeastern Reservoirs. Technical Report EL-95-14*. U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Kenney, B.C. 1990. *On the Dynamics of Phosphorus on Lake Systems*. Environment Canada, Ottawa.

Kenney, W.F., M.N. Waters, C.L. Schelske, and M. Brenner. 2002. Sediment records of phosphorus-driven shifts to phytoplankton dominance in shallow Florida lakes. *Journal of Paleolimnology* 27:367–377.

Lazzarino, J.K., R.W. Bachmann, M.V. Hoyer, and D.E. Canfield, Jr. 2009. Carbon dioxide supersaturation in Florida lakes. *Hydrobiologia* 627:169–180.

Legendre, P., and L. Legendre. 1998. *Numerical Ecology*. Second English Edition. Elsevier, Amsterdam.

Line, J.M., C.J.F. ter Braak, and H.J. Birks. 1994. WACALIB version 3.3 – A computer program to reconstruct environmental variables from fossil assemblages by weighted averaging and to derive sample-specific errors of prediction. *Journal of Paleolimnology* 10:147–152.

Lowe, E.F., L.E. Battoe, M. Coveney, and D. Stites. 1999. Setting water quality goals for restoration of Lake Apopka. *Lake and Reservoir Management.*15(2):103–120.

Lowe, E.F., L.E. Battoe, M.F. Coveney, C.L. Schelske, K.E. Havens, E.R. Marzolf, and K.R. Reddy. 2001. The restoration of Lake Apopka in relation to alternative stable states: an alternative view to that of Bachmann et al. (1999). *Hydrobiologia* 448:11–18.

US EPA ARCHIVE DOCUMENT

US EPA ARCHIVE DOCUMENT

Lowe, E., L. Battoe, J.C. Hendrickson, M. Coveney, R. Fulton, E. Marzolf, S. Winkler and J. Di. 2009. A Morphoedaphic Index to Predict Natural Background and Designated Use Failure Threshold Phosphorus and Chlorophyll Concentrations in Florida Lakes. SJRWMD Draft Report.

Mankin, K.R., S.H. Wang, J.K. Koelliker, D.G. Huggins, and F. de Noyelles. 2003. Watershed-lake water quality modeling: Verification and application. *Journal of Soil and Water Conservation* 58(4):188–197.

Muhammetoglu, A., H. Muhammetoglu, S. Oktas, L. Ozgokcen, and S. Soyupak. 2005. Impact Assessment of Different Management Scenarios on Water Quality of Porsuk River and Dam System – Turkey. *Water Resources Management* (2005) 19:199–210.

NAS (National Academy of Science). 1969. *Eutrophication: Causes, Consequences, Correctives*. National Academy of Science, Washington, DC.

OECD (Organisation for Economic Development and Co-Operation). 1982. *Eutrophication of Waters*. Monitoring, assessment and control. Organisation for Economic Development and Co-Operation, Paris.

Oglesby, R.T. 1977. Relationships of fish yield to lake phytoplankton standing crop, production, and morphoedaphic factors. *Journal of the Fisheries Research Board of Canada* 34:2271–2279.

Osborne, J.W. 2000. Advantages of hierarchical linear modeling. *Research Evaluation* 7(1). <http://pareonline.net/getvn.asp?v=7&n=1>. Accessed October 2009.

Paerl, H.W. 1988. Nuisance phytoplankton blooms in coastal, estuarine, and inland waters. *Limnology and Oceanography* 33:823–847.

Paerl, H.W. 2009. Controlling Eutrophication along the Freshwater–Marine Continuum: Dual Nutrient (N and P) Reductions are Essential. *Estuaries and Coasts* 32:593–601.

Pauer, J.J., K.W. Taunt, W. Melendez, R.G. Kreis, Jr., and A.M. Anstead. 2007. Resurrection of the Lake Michigan Eutrophication Model, MICH1. *Journal of Great Lakes Research* 33:554–565.

Petrus, K. 2007. TMDL Report. Nutrient TMDL for the Winter Haven Southern Chain of Lakes (WBIDs 1521, 1521D, 1521E, 1521F, 1521G, 1521H, 1521J, 1521K). Florida Department of Environmental Protection, Division of Water Resource Management, Bureau of Watershed Management, Southwest District, Sarasota Bay–Peace–Myakka Basins.

Paul, M., and J. Gerritsen. 2002. *Nutrient Criteria for Florida Lakes: A Comparison of Approaches*. Tetra Tech Inc., Owings Mills, MD.

Paul, M., and J. Gerritsen. 2003. *Development of Florida Lake Nutrient Criteria: Summary and Synthesis*. Tetra Tech Inc., Owings Mills, MD.

Price, P.N., A.V. Nero, and A. Gelman. 1996. Bayesian prediction of mean indoor radon concentrations for Minnesota counties. *Health Physics* 71:922–936.

Reckhow, K.H. 1999. Water quality prediction and probability network models. *Canadian Journal of Fisheries and Aquatic Sciences* 56(7):1150–1158.

Riedinger-Whitmore, M., T. Whitmore, J. Smoak, M. Brenner, A. Moore, J. Curtis, and C.L. Schelske. (2005). Cyanobacterial proliferation is a recent response to eutrophication in many Florida lakes: a paleolimnological assessment. *Lake and Reservoir Management* 21:423–435.

Robertson D.M., and W. J. Rose. 2008. Water Quality, Hydrology, and Simulated Response to Changes in Phosphorus Loading of Butternut Lake, Price and Ashland Counties, Wisconsin, with Special Emphasis on the Effects of Internal Phosphorus Loading in a Polymictic Lake. USGS Scientific Investigations Report 2008–5053. <http://www.usgs.gov/pubprod>.

Rule, T. 2004. *Nutrient Criteria in Lakes and Reservoirs: Maryland's Approach*. Maryland Department of the Environment, Watershed Modeling Division, TMDL Development and Application Program, Annapolis, MD.

Ryder, R.A., S.R. Kerr, K.H. Loftus, and H.A. Regier. 1974. The morphodeaphic index, a fish yield estimator—review and evaluation. *Journal of the Fisheries Research Board of Canada* 31:663–688.

Salas, H.J., and P. Martino. 1991. A simplified phosphorus trophic state model for warm-water tropical lakes. *Water Research* 25(3):341–350.

Saunders, J. 2009. *Colorado Lake Nutrient Criteria-Update*. Colorado Department of Public Health and Environment, Water Quality Control Division, Standards Unit, Denver, CO.

Scheffer, M.S., S. Hosper, M.L. Meijer, B. Moss, and E. Jeppesen. 1993. Alternative equilibria in shallow lakes. *Trends in Ecology and Evolution* 8:275–279.

Scheffer, M., S. Carpenter, J.A. Foley, C. Folke, and B. Walker. 2001. Catastrophic shifts in ecosystems. *Nature* 413:591–596.

Scheffer, M., S. Szabo, A. Gragnani, E.H. van Nes, S. Rinaldi, N. Kautsky, J. Norberg, R.M.M. Roijackers, and R.J.M. Franken. 2003. Floating plant dominance as a stable state. *Proceedings of the National Academy of Sciences* 100(7):4040–4045.

Schindler, D.W. 1974. Eutrophication and recovery in experimental lakes: Implications for lake management. *Science* 184: 897-899.

Schindler, D.W., H. Kling, R.V. Schmidt, J. Prokopowich, V.E. Frost, R. A. Reid, and M.Capel.1973. Eutrophication of Lake 227 by addition of phosphate and nitrate: The second, third, and fourth years of enrichment 1970, 1971, and 1972. *Journal of the Fishery Research Board of Canada* 30:1415–1440.

US EPA ARCHIVE DOCUMENT

Shannon, E.E., and P.L. Brezonik. 1972. Limnological characteristics of north and central Florida lakes. *Limnology and Oceanography* 17:97–110.

Shelley, Z., D. Gilbert, K. Petrus. 2005. Dissolved Oxygen and Nutrient TMDLs for Lake Hancock and Lower Saddle Creek (WBID 1623L & 1623K). Florida Department Of Environmental Protection, Division of Water Resource Management, Bureau of Watershed Management, Southwest District, Lake Hancock Basin, Peace River Planning Unit. <http://www.dep.state.fl.us/water/tmdl>.

Smeltzer, E. and S. Quinn. 1996. A phosphorus budget, model, and load reduction strategy for Lake Champlain. *Lake and Reservoir Management* 12(3):381–393.

Smith, V.H. 2006. Responses of estuarine and coastal marine phytoplankton to nitrogen and phosphorus enrichment. *Limnology and Oceanography* 51:377–384.

Smith, V.H., G.D. Tilman, and J.C. Nekola. 1999. Eutrophication: impacts of excess nutrient inputs on freshwater, marine, and terrestrial ecosystems. *Environmental Pollution* 100:179-196.

Smith, V.H., S.B. Joye, and R.W. Howarth. 2006. Eutrophication of freshwater and marine ecosystems. *Limnology and Oceanography* 51:351–355.

Søndergaard, M., E. Jeppesen, J.P. Jensen, and S.L. Amsinck. 2005. Water framework directive: Ecological classification of Danish lakes. *Journal of Applied Ecology* 42:616–629.

Stansbury, J., and D.M. Admiraal. 2004. Modeling to evaluate macrophyte induced impacts to dissolved oxygen in a tailwater reservoir. *Journal of American Water Resources Association* 40(6):1483–1497.

Stauffer, R.E., and D.E. Canfield, Jr. 1992. Hydrology and alkalinity regulation of soft Florida waters: An integrated assessment. *Water Resources Research* 28:1631–1648.

Steinman, A.D., and B.H. Rosen. 2000. Lotic-lentic linkages associated with Lake Okeechobee, Florida. *Journal of the North American Benthological Society* 19(4): 733–741.

U.S. EPA (U.S. Environmental Protection Agency). 1998. *Lake and Reservoir Bioassessment and Biocriteria. Technical Guidance Manual*. EPA 841-B-98-007. U.S. Environmental Protection Agency, Office of Water, Washington, DC.

U.S. EPA (U.S. Environmental Protection Agency). 2000. *Nutrient Criteria Technical Guidance Manual: Lakes and Reservoirs*. EPA-822-B-00-001. U.S. Environmental Protection Agency, Office of Water, Washington, DC.

U.S. EPA (U.S. Environmental Protection Agency). 2007. *Hypoxia in the Northern Gulf of Mexico: An Update by the EPA Science Advisory Board*. EPA-SAB-08-003. EPA Science Advisory Board, Washington, DC.

US EPA ARCHIVE DOCUMENT

US EPA ARCHIVE DOCUMENT

U.S. EPA (U.S. Environmental Protection Agency). 2008. *Nutrient Criteria Technical Guidance Manual: Wetlands*. EPA 822-R-07-004.U.S. Environmental Protection Agency, Office of Water, Washington, DC.

U.S. EPA (U.S. Environmental Protection Agency). 2009. *Proposed Methods and Approaches for Developing Numeric Nutrient Criteria for Florida's Inland Waters*. U.S. Environmental Protection Agency, Office of Water, Office of Science and Technology, Health and Ecological Criteria Division, Washington, DC.

U.S. EPA (U.S. Environmental Protection Agency). 2009. *Modeling Report, Sawgrass Lake, WBID 23981, Nutrients and DO*. U.S. Environmental Protection Agency, Region 4, Atlanta, GA.

Vadeboncoeur, Y., M.J. Vander Zanden, and D.M. Lodge. 2002. Putting the lake back together: Reintegrating benthic pathways into lake food web models. *Bioscience* 52(1):44–54.

Vestergaard, O. and K. Sand-Jensen. 2000. Aquatic macrophyte richness in Danish lakes in relation to alkalinity, transparency, and lake area. *Canadian Journal of Fisheries and Aquatic Sciences* 57:2022–2031.

Vighi, M., and G. Chiaudani. 1985. A simple method to estimate lake phosphorus concentrations resulting from natural background loadings. *Water Research* 19(8):987–991.

Vollenweider, R.A. 1968. Scientific Fundamentals of the Eutrophication of Lakes and Flowing Waters, With Particular Reference to Nitrogen and Phosphorus as Factors in Eutrophication (Tech Rep DAS/CS/68.27, OECD, Paris).

Vollenweider, R.A. 1975. Input-output models with special reference to the phosphorus loading concept in limnology. *Schweizerische Zeitschrift Fur Hydrologie* (Swiss Journal of Hydrology) 37:53–84.

Vollenweider, R.A. 1976. Advances in defining critical loading levels for phosphorus in lake management. *Journal Memorie dell'Istituto Italiano di Idrobiologia* 33:53–83.

Walker, W.W., Jr. 1984. Statistical bases for mean chl-a criteria. *Lake and Reservoir Management* 2:57–62.

Walker, W.W. 1981. *Empirical Methods for Predicting Eutrophication in Impoundments; Report 1, Phase I: Data Base Development.* Technical Report E–81–9, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Walker, W.W. 1982. *Empirical Methods for Predicting Eutrophication in Impoundments; Report 2, Phase II: Model Testing.* Technical Report E–81–9, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Walker, W.W. 1985. *Empirical Methods for Predicting Eutrophication in Impoundments; Report 3, Phase II: Model Refinements*. Technical Report E-81-9. U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Walker, W.W. 1987. Empirical Methods for Predicting Eutrophication in Impoundments; Report 4, Phase III: Applications Manual. Technical Report E-81-9. U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.; Walker, W.W., 1999. *Simplified Procedures for Eutrophication Assessment and Prediction: User Manual*; Instruction Report W-96-2, U.S. Army Corps of Engineers Waterways Experiment Station, Vicksburg, MS.

Wang, X.-D., S.-L. Liu, S.-S. Zhang, and J.-W. Chen. 2009. Improvement of WASP Eutrophication Model in Baiyangdian Water Area. *Huanjing Kexue yu Jiazhu* 32, (10):19-24.

Wetzel, R.G. 1975. *Limnology*. W.B. Saunders Company, Philadelphia, PA.

Whitmore, T. and M. Brenner. 2002. *Paleologic Characterization of Pre-disturbance Water Quality Conditions in EPA Defined Florida Lake Regions*. University of Florida, Department of Fisheries and Aquatic Sciences, Gainesville, FL.

WHO (World Health Organization). 1999. Toxic *Cyanobacteria in Water: A Guide to Their Public Health Consequences, Monitoring and Management*, eds. Chorus, I. and J. Bartram. ISBN 0-419-23930-8. World Health Organization, Geneva, Switzerland.

WHO (World Health Organization). 2003. *Guidelines for Safe Recreational Water Environments, Volume 1: Coastal and Fresh Waters*. World Health Organization, Geneva, Switzerland. <http://www.who.int/water_sanitation_health/bathing/srwe1/en/>. Accessed January 2010.

Wilton, T. 2008. *Nutrient Criteria and its Connection to Local Water Quality*. Iowa Department of Natural Resources, Des Moines, IA.

Winfield, I.J. 2004. Fish in the littoral zone: ecology, threats and management. *Limnologica* 34:124–131.

Wool, T.A., R.B. Ambrose, J.L. Martin, and E.A. Comer. 2001. Water Quality Analysis Simulation Program (WASP), Version 6.0, Draft: User's Manual. U.S. Environmental Protection Agency, Region 4, Atlanta, GA.

Yoko, S., R. Axler, M. McDonald, and D. Wilcox. 1997. Recovery of a mine pit lake from aquacultural phosphorus enrichment: model predictions and mechanisms. *Ecological Engineering* 8(3):195-218.

Zar, J.H. 1996. *Biostatistical Analysis*. Third Edition. Prentice-Hall.

US EPA ARCHIVE DOCUMENT

Chapter 3: Derivation of EPA's Numeric Nutrient Criteria for Springs

3.1 Introduction

Springs and their associated spring runs are a unique class of aquatic ecosystem that are highly treasured for their biological, economic, aesthetic, and recreational value. Because they primarily originate and are maintained by groundwater, most springs have water that is transparent and rich with dissolved ions. The latter is due to prolonged contact with subterranean limestone (Brown et al. 2008). Globally, the largest number of springs (approximately 600 to 700) occurs in Florida (Scott et al. 2004). Springs are often classified based on their flow rate, which ranges from more than 2,800 L/sec (first magnitude) to less than 0.47 L/sec (eighth magnitude). Many of the larger spring ecosystems in Florida have likely been in existence since the end of the last major ice age—approximately 15,000 to 30,000 years ago (Martin 1966; Munch et al. 2006). During this period, plant and animal communities have evolved to become highly adapted to the unique water quality and conditions found in the springs and spring runs. Springs also represent an important resource for humans, such as by providing a variety of recreational purposes (Scott et al. 2002). For example, many spring boil areas (i.e., the limestone vent where the majority of aquifer water is discharged to the surface, sometimes in a turbulent manner) in Florida have been modified to facilitate swimming, recreation, and even "health spas." Correspondingly, many springs have suffered declines (generally) in their condition (e.g., up-rooting of vegetation, bank erosion, litter) due to visitation by ever-increasing numbers of people (Florida Springs Task Force 2000).

This chapter describes the background, methodology, and results of EPA's analysis for deriving the proposed numeric nutrient criteria, including duration and frequency components, for Florida springs.

3.2 Effects of Nutrients on Springs

A significant anthropogenic factor linked with adverse changes in spring ecosystems is the pollution of groundwater, principally with nitrate-nitrogen (NO_3-nitrogen, hereafter referred to as NO_3+NO_2)[7], resulting from human land use changes, cultural practices, and general population growth. The NO_3+NO_2 associated with urban and agricultural activities seep through soils and are transported to springs in groundwater that emerges at springs where they stimulate the growth of excess algae and noxious plants (Brown et al. 2008).

Excess algal and plant growth results in a variety of adverse effects in Florida springs. Common adverse effects include reduced habitat and food sources for native wildlife, excess organic carbon production, accelerated decomposition, and lowered substrate quality—all of which affect the overall health and aesthetics of Florida's springs (Brown et al. 2008).

There are also documented adverse effects on native aquatic vegetation. As benthic algal mats accumulate, they kill beneficial submerged aquatic vegetation (SAV) through direct smothering

[7] Nitrate nitrogen (NO_3-nitrogen) is a closely associated inorganic nitrogen compound to nitrite nitrogen (NO_2-nitrogen). Because of the analytical methods used to quantify these compounds, they are often expressed together as NO_3+NO_2, or NO_3-NO_2 . Nitrite concentrations, however, tend to comprise a small proportion of the combined NO_3+NO_2 concentration (Andrews 1994; Phelps 2004).

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or indirectly through shading (Dennison and Abal 1999; Doyle and Smart 1998). Once the native SAV is displaced, other non-native taxa such as *Hydrilla verticillata* (hydrilla) can re-colonize bare substrates, leading to other biological and ecological changes. Prior to wide-scale development of Florida springs and their watersheds, native SAV (e.g., *Sagittaria kurziana*, *Vallisneria americana*) dominated the underwater regions near most spring vents. However, within the past 20 to 30 years nuisance macroalgae species have proliferated and are now out-competing and replacing SAV (Florida Springs Task Force 2000). In some of the more extreme examples, such as Silver Springs and Weeki Wachee Springs, algal mat accumulations have become several feet thick. Historic records of *Lyngbya wollei* exist from Silver Spring (Pinowska et al. 2007a) and show that prior to the last 20 to 30 years there were no observations of nuisance *Lyngbya* growth. In a recent survey of 60 first- and second-magnitude springs in Florida, the most commonly observed algal taxa were filamentous mat-forming cyanobacteria of the genus *Lyngbya* and the xanthophyte *Vaucheria* (Stevenson et al. 2004). Stevenson and colleagues found that the majority of the Florida springs they studied had nuisance growths of algae. Heffernan et al. (2010) offered an alternative hypothesis for excessive macroalgal growth in springs asserting that decreased dissolved oxygen in spring discharges result in changes algal grazer community structure, which in turn result in greater dominance of macroalgae in springs.

In addition to the ecological effects, specific types of algal growth populations, stimulated by nutrient pollution, can have adverse health effects on animals and humans. As noted above, *Lyngbya wollei* occurs across many springs in Florida. *Lyngbya wollei* proliferation is especially problematic because more than 70 biologically active compounds have been isolated from this species, many of which are potentially toxic and/or carcinogenic to humans and can therefore inhibit the recreational use of the resource (Osborne et al. 2001). It can produce a variety of paralytic shellfish poisons (e.g., saxitoxin) and other toxins capable of producing dermatitis in humans (Carmichael et al. 1997; Onodera et al. 1997; Stewart et al. 2006; Teneva et al. 2003) and deaths in domestic and wild animals that consume algal mats (Edwards et al. 1992; Falconer 1999; Gugger et al. 2005; Saker et al. 1999).

3.3 Numeric Nutrient Criteria Development

3.3.1 Nitrate+Nitrite (NO_3+NO_2)

EPA recommended to states that numeric nutrient criteria should address both causal parameters, such as TN and TP, as well as response variables, such as chl-*a* and clarity (U.S. EPA 2000). In developing numeric nutrient criteria to protect Florida's springs, EPA focused on NO_3+NO_2—not TN, TP, chl-*a*, and clarity—because NO_3+NO_2 is the predominant form of nutrient pollution in springs, and is known to be a driver of many of the adverse effects observed in Florida's springs (Brown et al. 2008). The concentrations of NO_3+NO_2 in groundwater and springs have been increasing over time in Florida.

While there are a variety of sources of nitrogen in springs, there appears to be no geologic source of NO_3+NO_2 as well as limited biogeochemical processes to retain or remove these compounds once introduced into the groundwater (Brown et al. 2008). Thus, NO_3+NO_2 is transported through groundwater to spring discharge points as a conservative solute and comprises the majority of TN found in springs (Cohen 2008). The resulting NO_3+NO_2 loads to springs are

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drivers of a variety of effects, which adversely impact the ecology and uses of Florida's springs (Brown et al. 2008; Pinowska et al. 2007a, 2007b).

In contrast, there does not appear to be a trend of increasing phosphorus concentrations in spring discharges despite Florida's naturally-rich phosphorus geology. Although NO_3+NO_2 concentrations have increased significantly in most spring discharges in Florida, phosphorus concentrations have remained relatively constant or decreased over the past 50 years (Strong 2004). This suggests the sources of phosphorus in groundwater that feed springs, are less influenced by anthropogenic activities compared to NO_3+NO_2 (Brown et al. 2008). EPA found no indication that the existing levels of phosphorus, which are considered to be derived from natural sources, contribute to the adverse effects observed in Florida springs. EPA also found no evidence of excess planktonic algal growth, as indicated by water column algal or phytoplankton biomass (chl-*a*), in Florida springs. This is likely due to the short residence times (or high flushing rates) of water at spring discharges resulting in low phytoplankton chl-*a*. Tychoplankton, or algae originating as benthic or attached algae, may be a larger source of water column chl-*a* in springs (Cohen 2008; Knight and Notestein 2008a, 2008b). Tychoplankton results from the circumstantial sloughing of accumulated epiphytic and benthic autotrophic communities by the high-flow, turbulent water in springs.

Many of Florida's springs are noted for their high degree of transparency, a feature possible due to the low level of light absorbing elements such as sediment, dissolved constituents (e.g., colored dissolved organic matter), and water column algal biomass. Transparency is a concern in springs and Florida has an existing criterion for water transparency in springs, which states, "the depth to the Light Compensation Point for photosynthetic activity shall not be reduced by more than 10% as compared to the natural background value" (Rule 62-302.530(67) F.A.C.).

Historically, natural background NO_3+NO_2 concentrations in spring discharges are thought to have been 0.05 mg/L or less (Maddox et al. 1992), which is sufficiently low to restrict the growth of native algae and vegetation under natural conditions. As indicated previously, the concentrations of NO_3+NO_2 in groundwater have increased in Florida over the last several decades as a result of human land use changes, cultural practices (e.g., such as fertilizer use and fossil fuel combustion), and general population growth (Florida Springs Task Force 2000). Increasing human populations have altered the global nitrogen cycle and other biogeochemical cycles through land use changes, fertilizer use, fossil fuel combustion, and other pathways (Vitousek et al.1997). Florida's karst region has experienced unprecedented population growth and changes in land use over the past several decades, with a consequential transfer of nutrients to the relatively unprotected groundwater. Katz et al. (1999) used isotopic analyses to show that substantial portions of the NO_3+NO_2 found in the Upper Floridan Aquifer and in spring discharges are derived from anthropogenic activities such as fertilizer application for agriculture and residential uses, livestock waste, and human waste. Figure 3-1 shows the changes in NO_3 concentration in Weeki Wachee Spring discharge as related to the population increase in Hernando County, Florida. The spring NO_3 concentrations follow a pattern very similar to the population curve with a 10 to 15 year lag. The lag period between changes on the land surface and the subsequent effect on spring discharges is expected because groundwater dating of water emerging from springs suggest that on average, it has spent between 10 and 30 years in the subsurface. However, these studies have also shown that a significant portion of water (30% to

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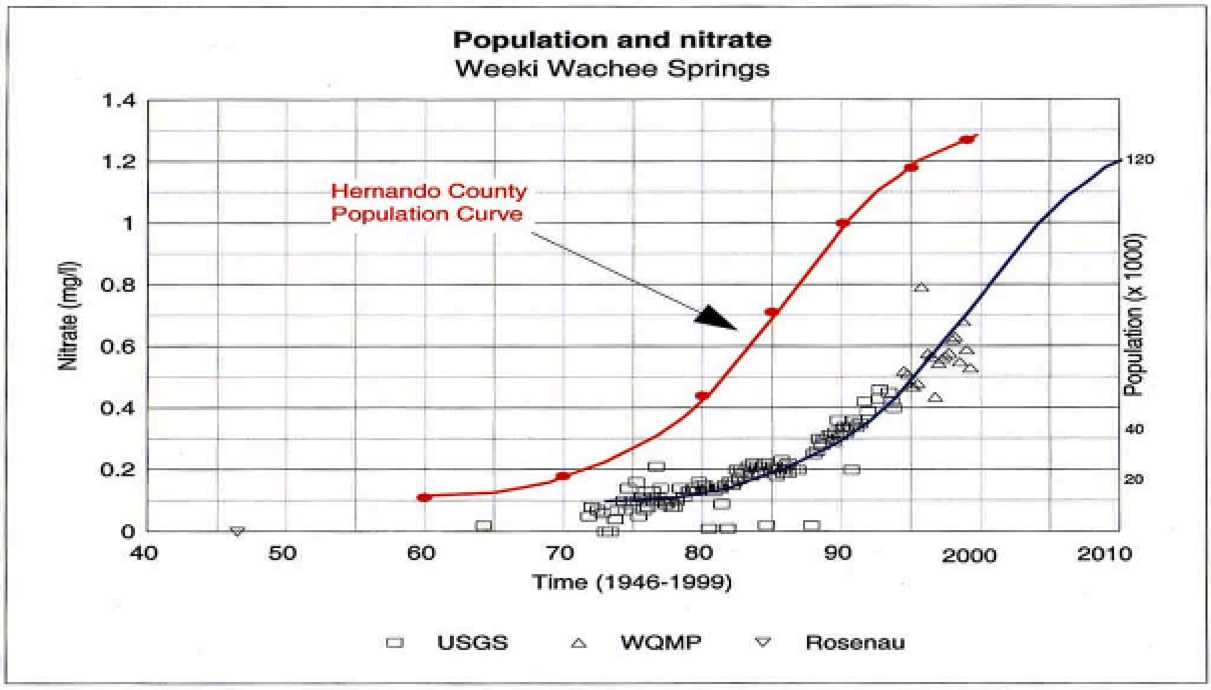
70%) has residence times less than 4 years and that the relative age contributions vary significantly between springs, depending on the characteristics of the recharge area of the spring.



Source: Florida Springs Task Force (2000).

Figure 3-1. Changes in NO_3 concentration in Weeki Wachee Spring discharge and population of Hernando County, Florida.
Note that NO_3 concentrations follow a pattern very similar to the population curve with a 10- to 15-year lag.

Of 125 springs sampled by the Florida Geological Survey in 2001 and 2002, 52 (42%) had NO_3+NO_2 concentrations exceeding 0.50 mg/L, and 30 (24%) had concentrations greater than 1.0 mg/L (Scott et al. 2004). Over 40% of the springs sampled had at least a tenfold increase in NO_3+NO_2 concentrations above background and approximately one-quarter of the springs demonstrated at least a 20-fold increase. Similarly, a recent evaluation of water quality in 13 first-magnitude springs shows that mean NO_3+NO_2 levels increased from 0.05 mg/L to 0.9 mg/L between 1970 and 2002 (Scott et al. 2004; Figure 3-2). Overall, these data suggest that NO_3+NO_2 concentrations in many spring discharges have increased from 10- to 350-fold over the past 50 years, with the level of increase closely correlated with the anthropogenic activity and land-use changes within the recharge area of the spring.

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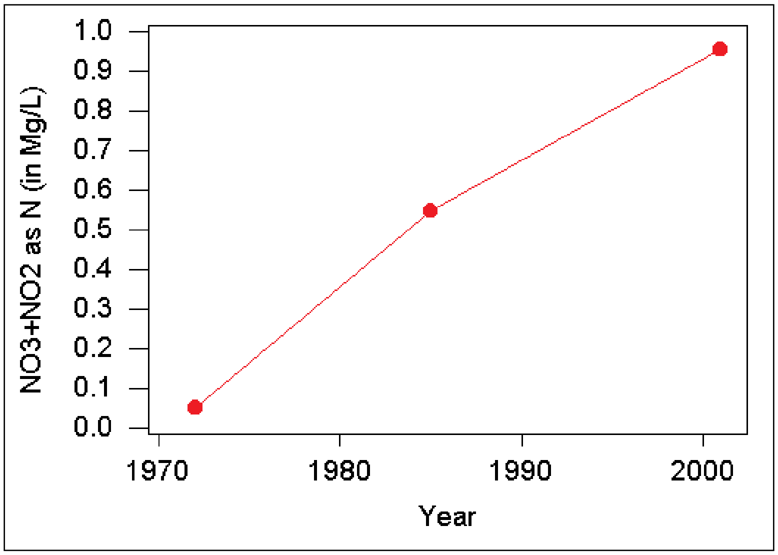

Source: Scott et al. (2004).

Figure 3-2. NO_3+NO_2 concentrations in discharges from 13 selected first-magnitude springs (Alexander, Chassahowitzka Main, Fanning, Ichetucknee Main, Jackson Blue, Madison Blue, Manatee, Rainbow Group composite, Silver Main, Silver Glen, Volusia Blue, Wakulla, and Wacissa #2 Springs) between the 1970s and the early 2000s.

As a result of the increased NO_3+NO_2 levels in groundwater and spring discharges, downstream NO_3+NO_2 loads are also increasing rapidly in many watersheds. For example, the NO_3+NO_2 concentrations of several springs in the Suwannee River Basin have increased over the past 40 years from less than 0.1 mg/L to more than 5 mg/L (Hornsby and Ceryak 2000) with the NO_3+NO_2-enriched spring discharge resulting in a 2- to 3-fold increase in the level of NO_3+NO_2 exported to the Gulf of Mexico by the Suwannee River. Consequently, the Suwannee Sound has experienced phytoplankton blooms in excess of 11 µg/L of chl-*a*, resulting in identification as an impaired water.

3.3.2 Nitrate+Nitrite Criteria Derivation

EPA reviewed various lines of evidence for developing NO_3+NO_2 criteria—primarily from previously published studies of Florida springs following the process outlined in EPA's analytical plan (Figure 3-3). EPA also reviewed the scientific evidence presented in support of FDEP's proposed springs criteria (FDEP 2009). EPA relied upon stressor-response approaches as the lines of evidence to support a criterion magnitude of 0.35 mg/L NO_3+NO_2. They include stressor-response analysis from controlled, laboratory dosing experiments with NO_3+NO_2, and stressor-response analysis from field surveys of biological communities and nutrient levels in Florida springs. Laboratory experiments can provide quantitative information about the response of algae (e.g., algal growth response) to varying nutrient concentrations under controlled conditions. Laboratory experiments with algae, however, do not include all the complexities and ecological processes that affect the response to nutrients in natural waterbodies (e.g., variable flow, light, and grazer conditions). Nonetheless, laboratory-based studies provide insights into the nutrient concentrations at which specific algal responses occur—particularly if these studies can be compared to and corroborated by observations and analyses of algal responses from field-based studies.

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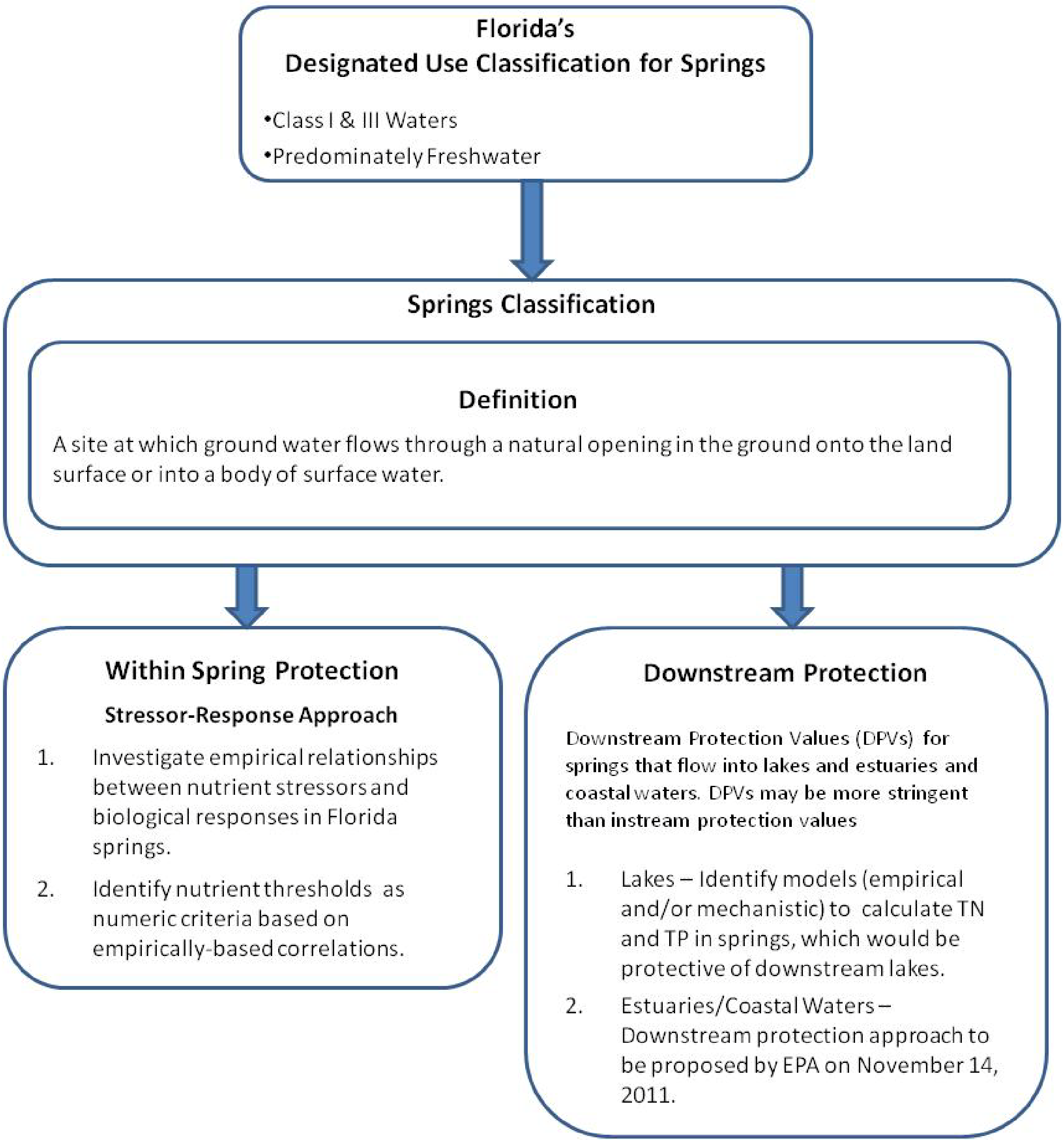

Figure 3-3. Analytical plan of EPA's numeric nutrient criteria development process for Florida's springs.

The first line of evidence EPA considered is a series of laboratory-based studies with macroalgae and nutrients by Pinowska et al. (2007b), summarized by Stevenson et al. (2007), which were conducted on behalf of FDEP. Pinowska et al. (2007b) examined the growth response of *Lyngbya wollei* in NO_3-dosed raceways. Their results showed that the biomass of small *Lyngbya wollei* mats approached maximum levels at NO_3 concentrations from 0.519–0.546 mg/L. In similar studies using *Vaucheria*, algal growth rates were low at NO_3 concentrations below 0.069 mg/L and increased substantially from 0.069 to 0.644 mg/L. Further growth rate increases

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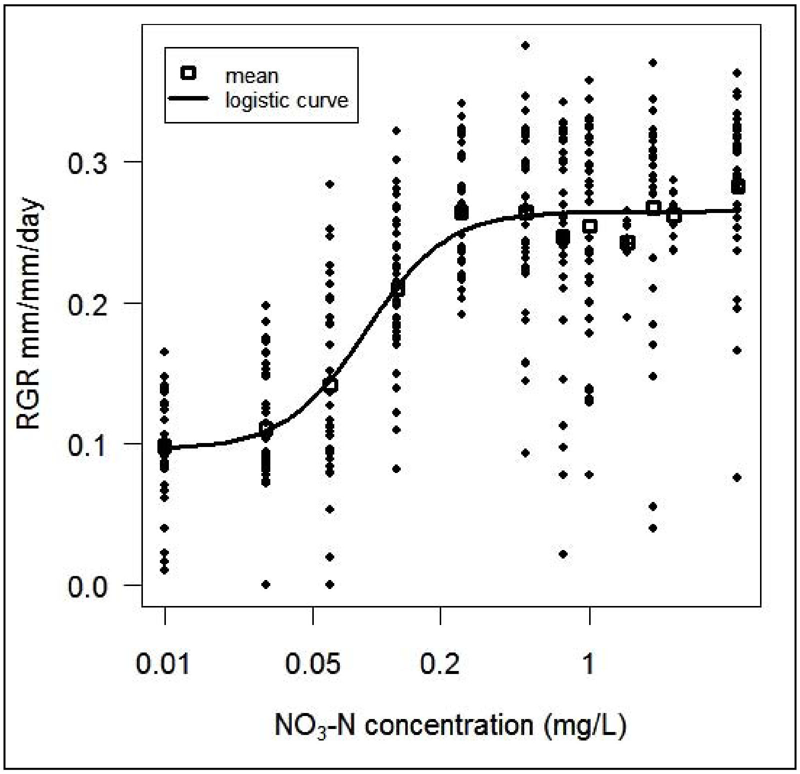
at NO_3 concentrations above 0.644 mg/L were minimal. Pinowska and colleagues also conducted small-scale experiments with microcentrifuge tube microcosms to evaluate the growth response of individual macroalgal filaments to specific doses of NO_3. Their results also showed that the minimum growth rate of *Lyngbya wollei* occurred at NO_3 concentrations below 0.034 mg/L. *Lyngbya wollei* growth rates increased rapidly between 0.034 to 0.230 mg/L NO_3 and reached maximum growth rates at NO_3 concentrations above 0.230 mg/L (Figure 3-4).



Source: Pinowska et al. (2007b).

Figure 3-4. *Lyngbya wollei* growth rates at various nitrate concentrations in microcentrifuge tubes.

Stevenson et al. (2007) attributed the differences in results between the raceway and microcentrifuge tube experiments to the differences in the scale of the experiments. Accurate control of nutrient levels was possible in the microcentrifuge tube microcosms using individual macroalgal filaments. In the larger-scale raceways using small algal mats, nutrient depletion was possible and because it could not be accounted for, resulting in a higher estimate of regulating NO_3 concentrations. Stevenson et al. (2007) recommended using the ED_{90} (i.e., NO_3+NO_2 concentration that produces 90% of the maximum growth) determined from the controlled microcentrifuge tube experiments as a preliminary NO_3 criterion that could be subsequently refined using additional information. The best estimates for the NO_3 ED_{90}s determined from the laboratory experiments were 0.230 mg/L for *Lyngbya wollei* and 0.261 mg/L for *Vaucheria* spp., respectively. Therefore, EPA considered 0.23 to 0.26 mg/L NO_3+NO_2 observed by Pinowska et al. (2007b) as a lower bound for a criterion in Florida springs.

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The second line of evidence EPA considered was based on the numerous surveys of springs that were conducted to demonstrate the cause and effect relationships between elevated nutrient concentrations and macroalgal growth. The benefit of using results of field surveys for nutrient criteria development is the direct applicability of observed nutrient concentrations and biological responses. In a survey of Florida springs, macroalgae were found at 59 of the 60 sampled sites, and an average of 50% of the spring bottoms were covered by macroalgae; the thickness of the macroalgal mats was commonly 0.5 m or more and as thick as 2 m in one spring boil (Stevenson et al. 2004). Although *Lyngbya* and *Vaucheria* spp. were the two most common taxa that occurred in extensive growths in the studied springs, a total of 23 different macroalgal taxa were observed in the spring survey.

Other surveys showed the abundance of *Vaucheria* species within the springs was positively correlated to nitrogen concentrations (Pinowska et al. 2007a). Nonlinear models of *Vaucheria* percent cover and thickness along TN and NO_3+NO_2 gradients explained substantially more variation than a linear model, with a threshold in *Vaucheria* response at 0.454 mg/L as NO_3+NO_2 (0.591 mg/L as TN). These researchers observed excessive growth and cover of *Vaucheria* spp. were found at sites with NO_3+NO_2 concentrations at or above the 0.454 mg/L threshold. Conversely, they observed less *Vaucheria* spp. abundance at sites with lower NO_3+NO_2 levels. EPA considers the excessive growth of macroalgae observed by Pinowska et al. (2007a) as an example of the imbalance of the natural biological communities stated in Florida's narrative nutrient criteria. Therefore, EPA considered 0.45 mg/L NO_3+NO_2 observed by Pinowska et al. (2007a) as an upper bound for a NO_3+NO_2 criterion in Florida springs.

Analysis of field data from the Wekiva River and Rock Springs Run (Gao 2008) also indicated that excess algal growth was associated with comparable NO_3+NO_2 concentrations observed by Pinowska et al. (2007a). The Wekiva River and Rock Springs Run are highly spring-influenced systems that are on the Florida's impaired waters list due to evidence of an imbalance in aquatic flora characterized by excessive algal growth and lower ecosystem metabolic activities. Gao (2008) found that mean NO_3+NO_2 concentration in the Wekiva River and Rock Springs Run ranged between 0.600 and 0.700 mg/L, which was higher than levels found at nearby minimally disturbed reference sites with similar characteristics (Juniper and Alexander Springs). EPA noted that the mean NO_3+NO_2 concentrations observed in the Wekiva and Rock Springs Run are two to three-fold higher than the ED90s for *Lyngbya wollei* and *Vaucheria* calculated by Pinowska et al. (2007b) and about 50% higher than the NO_3+NO_2 concentrations Pinowska et al. (2007a) found to be associated with nuisance *Vaucheria* spp. growth.

In a closer examination for the Wekiva River and Rock Springs Run TMDL by Gao (2008), NO_3+NO_2 targets were derived using stressor-response approaches based on periphyton and water quality monitoring data collected from the Suwannee River and two tributaries—the Withlacoochee River and Santa Fe River (Hornsby et al. 2000)—were also influenced highly by spring outflows and subterranean discharge. Gao (2008) considered these data to be applicable to the Wekiva River and Rock Springs Run because the Suwannee River is heavily influenced by spring inflow and, in the absence of anthropogenic inputs, the algal communities would be expected to be generally similar in composition to those in the Wekiva River and Rock Springs Run. An analysis of the periphytometer data collected during the period from 1990 through 1998 from 13 sites along the Suwannee River system (Hornsby et al. 2000) showed positive correlations between algal cell density and NO_3+NO_2 concentration, as well as periphyton

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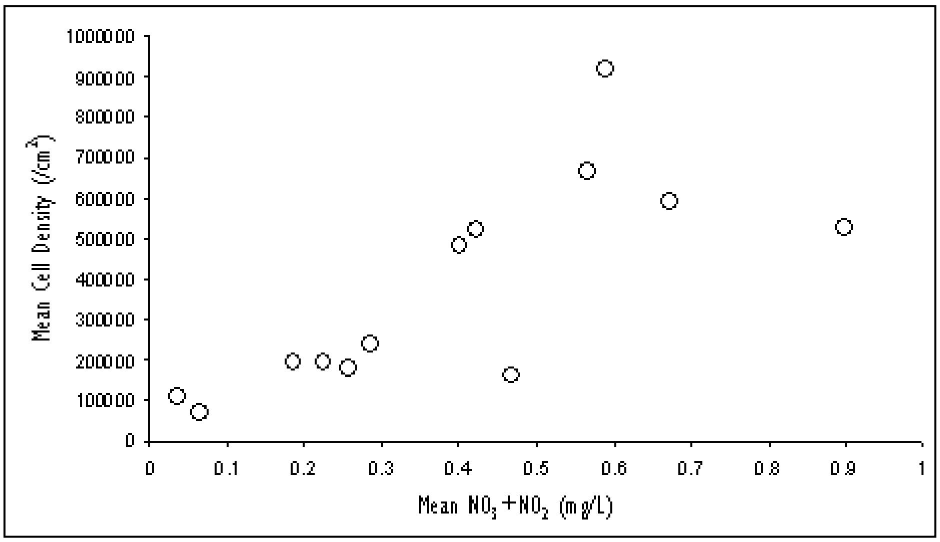
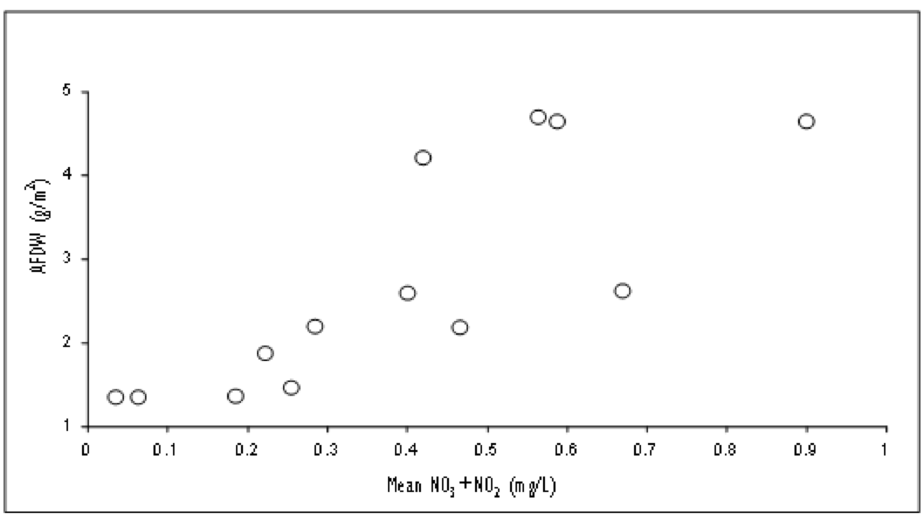
biomass and NO_3+NO_2 concentration (Niu 2007). The functional relationships of cell density versus NO_3+NO_2 concentration and periphyton biomass (represented as ash-free dry weight, or AFDW) versus NO_3+NO_2 concentration are shown in Figure 3-5 and Figure 3-6, respectively. Data were collected by Hornsby et al. (2000) and reported in Mattson et al. (2006) and Niu (2007). Note that the data presented in these figures represent long-term average biomass, cell densities, and NO_3+NO_2 concentrations at the stations across the Suwannee River system (Niu 2007).



Source: Mattson et al. (2006).

Figure 3-5. Relationship between mean NO_3+NO_2 concentration and mean periphyton cell density from sampling sites on the Suwannee, Santa Fe, and Withlacoochee Rivers.



Source: Mattson et al. (2006).

Figure 3-6. Relationship between mean NO_3+NO_2 concentration and mean periphyton biomass from sampling sites on the Suwannee, Santa Fe, and Withlacoochee Rivers.

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Niu (2007) further evaluated the data using change point analysis to better define the NO_3+NO_2 concentration that might significantly impact the periphyton biomass and cell density. The change point analysis fitted a step function through observed data by examining the probability of each data point as the change point. For both periphyton cell density and periphyton biomass, change point step functions were shown to be the best model among the models tested, and which supports the use of change point analysis. For cell density and NO_3+NO_2 concentration, the change point step function identified two populations of sites. The first population had cell densities near 163,000 cells/cm^2 (p = 0.009), which was considered the baseline condition under which no significant NO_3+NO_2 impact was detected. The second population had cell densities near 616,000 cells/cm^2 (p = 0.0001), which was significantly elevated above the baseline condition. The change point analyses also indicated that the critical increase in mean algal cell density occurred as the mean NO_3+NO_2 concentration increased from 0.286 to 0.401 mg/L. Similarly, the change point analysis of periphyton biomass and NO_3+NO_2 concentration identified two populations of sites. The first population had periphyton biomass near 1.73 g/m^2 (p< 0.0001), which was considered the baseline condition under which no significant NO_3+NO_2 impact was detected. The second population had a higher algal biomass near 4.15 g/m^2 (p = 0.0001). The change point analyses also indicated that the critical increase in mean periphyton biomass occurred as the mean NO_3+NO_2 concentration increased from 0.401 to 0.420 mg/L (Niu 2007). This approach by Niu (2007) suggested to EPA that to prevent the periphyton cell density from increasing to the higher level, the numeric nutrient criteria for NO_3+NO_2 should be established below 0.40 to 0.42 mg/L NO_3+NO_2, which is consistent with the findings of Pinowska et al. (2007a).

Niu (2008) repeated the change point analysis using the original data in Niu (2007) combined with additional data collected through 2007 for the same 13 stations located along the Suwannee River. To account for any long-term temporal changes at a site, the period of record was divided into four periods. The average periphyton abundance and NO_3+NO_2 data for each period for each station were used. The results were similar to those obtained from the original analyses (Niu 2007) as described above. Niu (2008) found a change point at 0.441 mg/L NO_3+NO_2 for both periphyton cell density and biomass (Figure 3-7 and Figure 3-8). EPA found that this analysis yielded a concentration that is similar to the threshold of 0.45 mg/L NO_3+NO_2 for *Vaucheria* spp. observed in the surveys of Pinowska et al. (2007a).

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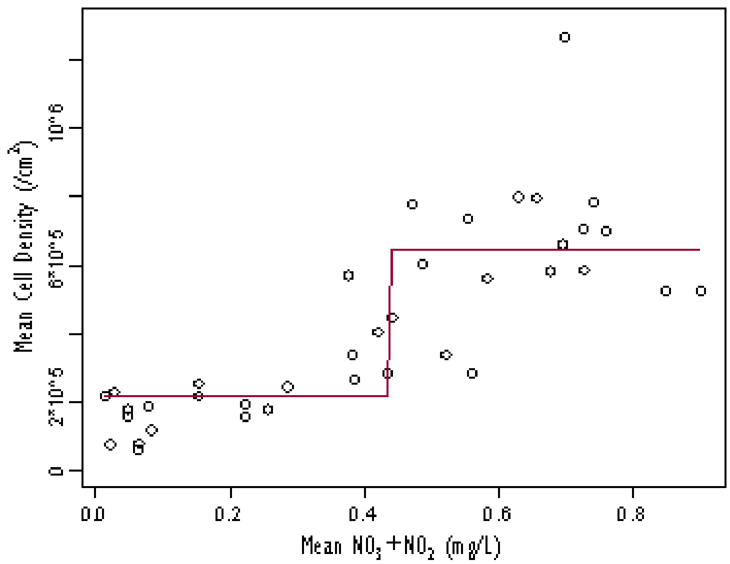
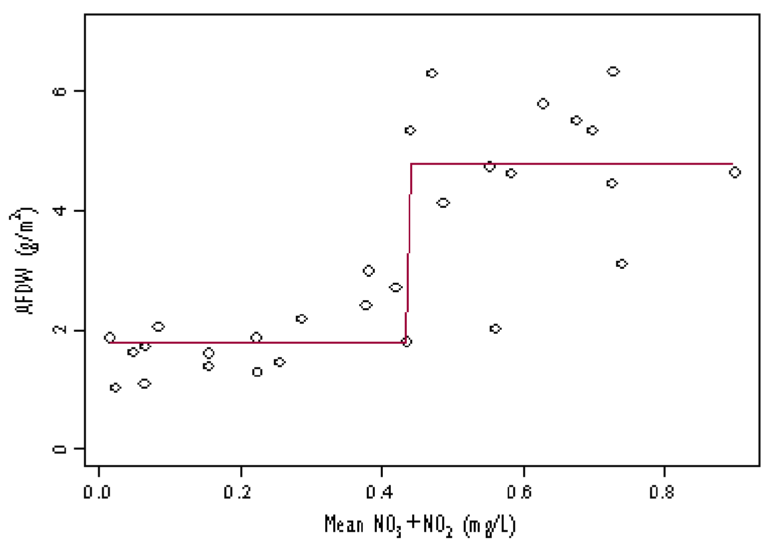


Source: Hallas and Magley (2008).

Figure 3-7. Change point analysis for data from the 13 stations at the Suwannee River system (mean cell density vs. mean NO_3+NO_2).
Note that change point = 0.441 mg/L NO_3+NO_2. The 95% confidence interval for the change point based on 1000 bootstrapping samples is 0.378–0.629 mg/L.



Source: Hallas and Magley (2008).

Figure 3-8. Change point analysis for data from the 13 stations in the Suwannee River system (mean biomass vs. mean NO_3+NO_2).
Note that change point = 0.441 mg/L. The 95% confidence interval for the change point based on 1000 bootstrapping samples is 0.441–0.584 mg/L.

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EPA found evidence that indicated controlling NO_3+NO_2 concentrations in springs would result in a reduced frequency, intensity, and duration of nuisance macroalgal growth in springs and prevent biological imbalances. The laboratory-based experimental results from Pinowska et al. (2007b) indicated NO_3 concentrations less than 0.230 mg/L NO_3 and 0.261 mg/L NO_3 would prevent excess growth of *Lyngbya wollei* and *Vaucheria*, respectively. Analyses from periphyton field surveys of springs indicated that NO_3+NO_2 concentrations would need to be reduced below the observed 0.454 mg/L threshold to reduce the nuisance abundance and cover of *Vaucheria* species in Florida springs (Pinowska et al. 2007a). Change point analyses by Niu (2007) and Niu (2008) indicated NO_3+NO_2 concentrations below 0.40 to 0.44 mg/L would prevent excess algal growth in springs. EPA found that the weight-of-evidence supported a NO_3+NO_2 criteria between these two lines of stressor-response evidence—at 0.350 mg/L NO_3+NO_2—to protect Florida's designated uses in springs. EPA believes a criterion at this concentration balances the uncertainty inherent in the translating ideal laboratory conditions to the field versus uncertainty inherent in estimating stressor-response relationships from field data.

3.4 Duration and Frequency for Springs Criteria

The algal responses in the laboratory experiments by Pinowska et al. (2007b) indicate that the biological response to NO_3+NO_2 can occur over the timescales of a month. A simple model developed by Stevenson et al. (2007) also indicates that significant changes in algal biomass could be achieved during a 1-month period by changes in algal growth rate. Although springs appear to be relatively stable habitats, their characteristics can vary over different time scales. Cohen (2008) (discussed in Brown et al. 2008) noted that the concentration of nitrate and other compounds in water discharging from springs is strongly correlated with discharge rate. This discharge is a mixture of recent and less recent water, with the recent water carrying a greater portion of the nitrate load. Because the more recent water is affected by variability in meteorological events, the discharge rates and nitrate concentration in spring vents can vary on short time frames (Cohen 2008). EPA also observed that nitrate concentration can vary on an inter-annual basis as well (Brown et al. 2008). To accurately capture this variability, EPA concluded that the most appropriate approach to characterizing NO_3+NO_2 in springs is over an annual averaging basis, or more specifically as an annual geometric mean NO_3+NO_2. For frequency of excursion, EPA considered the variable temporal responses of algae to NO_3+NO_2 in the various studies previously described and concluded that the springs criteria should not be exceeded more than once (as an annual geometric mean) over a three year period.

3.5 Summary

Florida springs and spring runs are unique and valuable resources. They are not only prized by humans for their inherent beauty, but also necessary to aquatic organisms for their combination of physical and chemical conditions. However, a documented increase of NO_3+NO_2 concentration in springs caused by human activities has caused a shift from systems dominated by SAV to systems dominated by attached algae, and which have impacted the integrity of springs. The criteria established for springs and spring runs are intended to prevent the further and extensive damage that will occur if trends in NO_3+NO_2 concentration observed in spring discharges are left unchecked.

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The analyses conducted by EPA to derive criteria for springs represent a Florida-specific effort that accounts for the available data and unique set of conditions influencing water quality, water quantity, and biology in springs. Human use of the land, climate patterns, and subterranean geology and hydrology, are all noted for the role they play in creating a system with historically low nitrogen levels, little assimilation capacity for excess nitrogen, and relatively low (when compared to other surface waters) concentrations of phosphorus. The best available scientific literature pertaining to nutrients and their role in springs highlighted the critical role of NO_3+NO_2 in springs—specifically the linkage between increased NO_3+NO_2 and increased algal growth and the weight of this literature was subsequently used to establish a criterion for NO_3+NO_2. Stressor response data from laboratory studies and field surveys informed the derivation of the final criteria magnitude of 0.35 mg/L NO_3+NO_2 for Florida springs. The duration and frequency reflects the intra- and inter-annual variability in NO_3+NO_2 observed in Florida springs, taking into account the observed biological responses to NO_3+NO_2.

3.6 References

Andrews, W.J. 1994. *Nitrate in Ground Water and Spring Water near Four Dairy Farms in North Florida, 1990-93*. U.S. Geological Survey Water-Resources Investigations Report 94-4162.

Brown, M.T., K.C. Reiss, M.J. Cohen, J.M. Evans, P.W. Inglett, K.S. Inglett, K.R. Reddy, T.K. Fraze, C.A. Jacoby, E.J. Phlips, R.L. Knight, S.K. Notestein, R.G. Hamann, and K.A. McKee. 2008. *Summary and Synthesis of the Available Literature on the Effects of Nutrients on Spring Organisms and Systems*. University of Florida, Gainesville, FL. <http://www.dep.state.fl.us/springs/reports/files/UF_SpringsNutrients_Report.pdf>. Accessed January 2010.

Cardellina, J.H., F.J. Marner, and R.E. Moore. 1979. Seaweed dermatitis: structure of *Lyngbya* toxin 16 A. *Science* 204:193–195.

Carmichael, W.W., W.R. Evans, Q.Q. Yin, P. Bell, and P. Moczyydlowski. 1997. Evidence for paralytic shellfish poisons in the freshwater cyanobacterium *Lyngbya wollei* (Farlow ex Gomont) comb. nov. *Applied and Environmental Microbiology* 63:3104–3110.

Cohen, M.J. 2008. Springshed Nutrient Loading, Transport and Transformations. Chapter 2 in *Summary and Synthesis of the Available Literature on the Effects of Nutrients on Spring Organisms and Systems*. Report prepared for the Florida Department of Environmental Protection, Tallahassee, FL.

Dennison,W.C. and E.G. Abal. 1999. *Moreton Bay Study: A Scientific Basis for the Healthy Waterways Campaign*. South East Queensland Regional Water Quality Management Strategy Team, Brisbane, Australia.

Doyle, R.D., and R.M. Smart. 1998. Competitive reduction of noxious *Lyngbya wollei* mats by rooted aquatic plants. *Aquatic Botany* 61:17–32.

US EPA ARCHIVE DOCUMENT

Duarte, C.M. 1995. Submerged aquatic vegetation in relation to different nutrient regimes. *Ophelia* 41:87–112.

Edwards, C., K.A. Beattie, C.M. Scrimgeour, and G.A. Codd. 1992. Identification of anatoxin-a in benthic cyanobacteria (blue-green algae) and in associated dog poisonings at Loch Insh, Scotland. *Toxicon* 30:1165–1175.

Falconer, I.R. 1999. An overview of problems caused by toxic blue-green algae (Cyanobacteria) in drinking and recreational water. *Environmental Toxicology* 14:5–12.

Florida Administrative Code. Chapter 62-302, *Surface Water Quality Standards*.

Florida Springs Task Force. 2000. *Florida's Springs: Strategies for Protection & Restoration*. Florida Springs Report. Florida Department of Environmental Protection, Tallahassee, FL. <http://www.dep.state.fl.us/springs/reports/index.htm>. Accessed January 2010.

Gao, X. 2008. *Nutrient TMDLs for the Wekiva River (WBIDs 2956, 2956A, and 2956C) and Rock Springs Run (WBID 2967).* Florida Department of Environmental Protection, Tallahassee, FL.

Gugger, M., S. Lenoir, C. Berger, A. Ledreux, J.C. Druart, J.F. Humbert, C. Guette, and C. Bernard. 2005. First report in a river in France of the benthic cyanobacterium *Phormidium favosum* producing anatoxin-a associated with dog neurotoxicosis. *Toxicon* 45:919–928.

Hallas, J.F. and W. Magley. 2008. Nutrient and Dissolved Oxygen TMDL for the Suwannee River, Santa Fe River, Manatee Springs (3422R), Fanning Springs (3422S),Branford Spring (3422J), Ruth Spring (3422L), Troy Spring (3422T), Royal Spring (3422U), and Falmouth Spring (3422Z). Florida Department of Environmental Protection, Tallahassee, FL. Hamill, K.D. 2001. Toxicity in benthic freshwater cyanobacteria (blue-green algae): First observations in New Zealand. *New Zealand Journal of Marine and Freshwater Research* 35:1057–1059.

Heffernan, J.B., Liebowitz, D.M., Frazer, T.K., Evans, J.M., and M.J. Cohen. 2010. Algal blooms and the nitrogen-enrichment hypothesis in Florida springs: evidence, alternatives, and adaptive management. *Ecological Society of America* 20:816–829.

Hornsby, D. and R. Ceryak. 2000. *Springs of the Aucilla, Coastal, and Waccasassa Basins in Florida*. Suwannee River Water Management District Technical Report WR 00-03. Suwannee River Water Management District, Live Oak, FL.

Hornsby, D., R.A. Mattson, and T. Mirti. 2000. *Surface Water Quality and Biological Monitoring.* Suwannee River Water Management District Technical Report WR-00-04. Suwannee River Water Management District, Live Oak, FL.

US EPA ARCHIVE DOCUMENT

Inglett, P.W., K.S. Inglett, and K.R. Reddy. 2008. Biogeochemical Processes and Implications for Nutrient Cycling. Chapter 3 in *Summary and Synthesis of the Available Literature on the Effects of Nutrients on Spring Organisms and Systems*. Report prepared for the Florida Department of Environmental Protection, Tallahassee, FL.

Katz, B.G., H.D. Hornsby, J.F. Bohlke, and M.F. Mokray. 1999. *Sources and Chronology of Nitrate Contamination in Spring Water*. U. S. Geological Survey Water-Resources Investigations Report 99–4252. Suwannee River Water Management District, Live Oak, FL.

Knight, R.L. and S.K. Notestein. 2008a. Effects of Nutrients on Spring Ecosystems. Chapter 6 in *Summary and Synthesis of the Available Literature on the Effects of Nutrients on Spring Organisms and Systems*. Report prepared for the Florida Department of Environmental Protection, Tallahassee, FL.

Knight, R.L. and S.K. Notestein. 2008b. Springs as Ecosystems. Chapter 1 in *Summary and Synthesis of the Available Literature on the Effects of Nutrients on Spring Organisms and Systems*. Report prepared for the Florida Department of Environmental Protection, Tallahassee, FL.

Maddox, G.L., J.M. Lloyd, T.M. Scott, S.B. Upchurch, and R. Copeland. 1992. *Florida's Groundwater Quality Monitoring Program – Background Hydrochemistry*. Special Publication 34. Florida Geological Survey, Tallahassee, FL.

Martin, R.A. 1966. *Eternal Spring: Man's 10,000 Years of History at Florida's Silver Springs*. Great Outdoors Press, Inc., St. Petersburg, FL.

Mattson, R.A., E.F. Lowe, C.L. Lippincott, D. Jian, and L. Battoe. 2006. *Wekiva River and Rock Springs Run Pollutant Load Reduction Goals*. St. Johns River Water Management District, Palatka, FL.

Munch, D.A., D.J. Toth, C. Huang, J.B. Davis, C.M. Fortich, W.L. Osburn, E.J. Phlips, E.L. Quinlan, M.S. Allen, M.J. Woods, P. Cooney, R.L. Knight, R.A. Clarke, and S.L. Knight. 2006. *Fifty-year Retrospective Study of the Ecology of Silver Springs, Florida*. Publication Number: SJ2007-SP4. St. Johns River Water Management District, Palatka, FL.

Mynderse, J.S., R.E. Moore, M. Kashiwagi, and T.R. Norton. 1977. Antileukemia activity in the Osillatoriaceae: isolation of debromoaplysiatoxin from *Lyngbya*. *Science* 196:538–540.

Niu, X. 2007. *Change Point Analysis of Suwannee River Algal Data*. Technical report prepared for the Florida Department of Environmental Protection, Tallahassee, FL.

Niu, X. 2008. *Change Point Analysis of the Suwannee River Algal Data: Change Point Analysis of Suwannee River Algal Data Based on an Updated Data Set*. Florida Department of Environmental Protection, Tallahassee, FL.

US EPA ARCHIVE DOCUMENT

Onodera, H., M. Satake, Y. Oshima, T. Yasumoto, and W.W. Carmichael. 1997. New saxitoxin analogues from the filamentous cyanobacterium *Lyngbya wollei*. *Natural Toxins* 5:146–151.

Osborne, N.J.T., P.M. Webb, and G.R. Shaw. 2001. The toxins of Lyngbya majuscula and their human and ecological health effects. *Environment International* 27:381–392.

Phelps, G.G. 2004. *Chemistry of Ground Water in the Silver Springs Basin, Florida, with an Emphasis on Nitrate*. U.S. Geological Survey Scientific Investigations Report 2004-5144.

Pinowska, A., R. J. Stevenson, J. O. Sickman, A. Albertin, and M. Anderson. 2007a. *Integrated Interpretation of Survey for Determining Nutrient Thresholds for Macroalgae in Florida Springs: Macroalgal Relationships to Water, Sediment and Macroalgae Nutrients, Diatom Indicators and Land Use*. Florida Department of Environmental Protection, Tallahassee, FL.

Pinowska, A., R. J. Stevenson, J. O. Sickman, A. Albertin, and M. Anderson. 2007b. *Integrated Interpretation of Survey and Experimental Approaches for Determining Nutrient Thresholds for Macroalgae in Florida Springs: Laboratory Experiments and Disturbance Study*. Florida Department of Environmental Protection, Tallahassee, FL.

Pittman. J.R., H.H. Hatzell, E.T. Oaksford. 1997. *Spring Contributions to Water Quantity and Nitrate Loads in the Suwannee River during Base Flow in July 1995*. U.S. Geological Survey, Water-Resources Investigations Report.

Rabalais, N. 2002. Nitrogen in aquatic ecosystems. *Ambio* 31:102–112.

Saker, M.L., A.D. Thomas, and J.H. Norton. 1999. Cattle mortality attributed to the toxic cyanobacterium *Cylindrospermopsis raciborskii* in an outback region of North Queensland. *Environmental Toxicology* 14:179–182.

Scott, T.M., G.H. Means, R.C. Means, and R.P. Meegan. 2002. *First Magnitude Springs of Florida*. Open File Report No. 85. Florida Geological Survey, Tallahassee, FL.

Scott, T.M., G.H. Means, R.P. Meegan, R.C. Means, S.B. Upchurch, R.E. Copeland, J. Jones, T. Roberts, and A. Willet. 2004. *Springs of Florida*. Bulletin No, 66. Florida Geological Survey, Tallahassee, FL.

Stevenson, R.J., A. Pinowska, and Y.K. Wang. 2004. *Ecological Condition of Algae and Nutrients in Florida Springs*. Florida Department of Environmental Protection, Tallahassee, FL.

Stevenson, R.J., A. Pinowska, A. Albertin, and J.O. Sickman. 2007. *Ecological Condition of Algae and Nutrients in Florida Springs: the Synthesis Report*. Prepared for the Florida Department of Environmental Protection, Tallahassee, FL.

US EPA ARCHIVE DOCUMENT

Stewart, I., I.M. Robertson, P.M. Webb, P.J. Schluter, and G.R. Shaw. 2006. Cutaneous hypersensitivity reactions to freshwater cyanobacteria–human volunteer studies. *BMC Dermatology.* 6:6.

Strong, W. 2004. Temporal Water Chemistry Trends within Individual Springs and within a Population of Florida Springs. M.S. Thesis, University of Florida, Gainesville FL.

Teneva, I., D. Asparuhova, B. Dzhambazov, R. Mladenov, and K. Schirmer. 2003. The freshwater cyanobacterium *Lyngbya* aerugineo-coerulea produces compounds toxic to mice and to mammalian and fish cells. *Environmental Toxicology* 18:9–20.

Van Dam, H., A. Mertens, and J. Sinkeldam. 1994. A coded checklist and ecological indicator values of freshwater diatoms from the Netherlands. *Netherlands Journal of Aquatic Ecology* 28:117–133.

Vitousek, P.M. J.D. Aber, R. W. Howarth, G.E. Likens, P.A. Matson, D.W. Schindler, W.H. Schlesinger, and D.G. Tilman. 1997. Human alteration of the global nitrogen cycle: Sources and consequences. *Ecological Applications* 7:737–750.

US EPA ARCHIVE DOCUMENT