Methods and Approaches for Deriving Numeric Criteria for Nitrogen/Phosphorus Pollution in Florida's Estuaries, Coastal Waters, and Southern Inland Flowing Waters

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Disclaimer

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Abbreviations and Acronyms

BMAP	Basin Management Action Plan
BMP	best management practice
BOD	biological oxygen demand
CAF	Coastal Assessment Framework
CBOD	carbonaceous biochemical oxygen demand
CDA	coastal drainage area
CDOM	colored dissolved organic matter
CE-QUAL-ICM	three-dimensional eutrophication model
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CFR	Code of Federal Regulations
CH3D	Curvilinear-grid Hydrodynamics 3d
Chl-a	chlorophyll *a*
Chl_{RS}-a	satellite-derived chlorophyll *a*
cm	centimeters
CUSUM	cumulative sum
CWA	Clean Water Act
DO	dissolved oxygen
DOC	dissolved organic carbon
DOM	dissolved organic matter
DOP	dissolved organic phosphorus
DPV	downstream protective values
DYNHYD	WASP hydrodynamics model
EAA	Everglades Agricultural Area
ECOHAB	Ecology and Oceanography of Harmful Algal Blooms project
ECOM-3D	Estuarine Coastal Ocean Model
EDA	estuarine drainage area
EFA	Everglades Forever Act
EFDC	Environmental Fluid Dynamics Code
EPA	United States Environmental Protection Agency
EPT	Ephemeroptera, Plecotera, and Trichoptera
ERA	environmental risk assessment
ESA	European Space Agency
EvPA	Everglades Protection Area
F.A.C.	Florida Administrative Code
FDA	fluvial drainage area
FDEP	Florida Department of Environmental Protection
FL	Florida
FWRI	Fish and Wildlife Research Institute
GIS	geographic information system
GLUT	Georgia Land Use Trends
GM	geometric mean

GQUAL	generalized quality constituent module
HAB	harmful algal bloom
HAB-OFS	Harmful Algal Bloom Operation Forecast System
HSPF	Hydrological Simulation Program—Fortran
HUC	hydrologic unit code
HYCOM	Hybrid Coordinate Ocean Model
IPV	instream protective value
IWR	Impaired Waters Rule
km	kilometers
k_{TN}	first-order decay rate for total nitrogen
k_{TP}	first-order decay rate for total phosphorus
L	liter
LDI	Landscape Development Intensity Index
LN	natural logarithm
$L_M(N)$	total nitrogen load limit
$L_M(P)$	total phosphorus load limit
LOESS	locally weighted scatterplot smoothing
log_{10}	logarithm, base 10
LSPC	Loading Simulation Program in C++
LTER	Long Term Ecological Research
MERIS	Medium Resolution Imaging Spectrometer
MGD	million gallons per day
MODIS	Moderate Resolution Imaging Spectroradiometer
N	nitrogen
NASA	National Aeronautics and Space Administration
NBOD	nitrogenous biochemical oxygen demand
NCDC	National Climatic Data Center
NEGOM	Northeastern Gulf of Mexico project
NH_3	ammonia
NHDPlus	National Hydrography Dataset Plus
NLCD	National Land Cover Data
NO_3-NO_2	nitrate+nitrite
NOAA	National Oceanic and Atmospheric Administration
NODC	National Oceanographic Data Center
NPDES	National Pollutant Discharge Elimination System
NPOESS	National Polar-orbiting Operational Environmental Satellite System
NPP	National Polar-orbiting Operational Environmental Satellite System Preparatory Project
P	phosphorus
PAR	photosynthetically active radiation
PLRG	pollutant load reduction goal
PLSM	Pollutant Load Screening Model
POM	Princeton Ocean Model
POP	particulate organic phosphorus
ppb	parts per billion
PQUAL	pervious quality constituent module

psu	practical salinity units
q_s	hydraulic loading
R^2	coefficient of regression
RCA	Row Column AESOP
RCRA	Resource Conservation and Recovery Act
RTAG	Regional Technical Assistance Group
SAB	Science Advisory Board
SAGT	USGS regional SPARROW model for the South Atlantic, eastern Gulf Coast and Tennessee River
SAV	submerged aquatic vegetation
SCI	stream condition index
SeaBASS	SeaWiFS Bio-optical Archive and Storage System
SeaDAS	SeaWiFS Data Analysis System
SeaWiFS	Sea-viewing Wide Field-of-view Sensor
SERC	Southeast Environmental Research Center
SOD	sediment oxygen demand
SPARROW	SPAtially Referenced Regression On Watershed attributes model
SPCC	Spill Prevention Control and Countermeasures
SSAC	Site-Specific Alternative Criteria
STA	stormwater treatment area
STATSGO	State Soil Geographic Database
TKN	total Kjeldhal nitrogen
TMDL	total maximum daily load
TN	total nitrogen
TP	total phosphorus
TSI	Trophic Status Index
USEPA	United States Environmental Protection Agency
USGS	United States Geological Survey
VIIRS	Infrared Imaging Radiometer Suite
WAFR	Wastewater Facility Regulation
WAMView	Watershed Assessment Model
WASP	Water Quality Analysis Simulation Program
WBID	waterbody identification number
WCA	Water Conservation Area
Z_c	colonization depth
τ_w	water residence time
µg	micrograms
‰	parts per thousand

Executive Summary

Excess inputs of nitrogen and phosphorus (nitrogen/phosphorus pollution) in surface waters can be harmful in aquatic ecosystems by directly producing excess plant and algal growth, and indirectly leading to reduced clarity, reduced oxygen levels as the algae and plants decompose, and decreased biodiversity. Primary sources of nitrogen and phosphorus to aquatic ecosystems include waste water and sewage effluent, atmospheric deposition, landfill leachate, fossil fuel combustion, and runoff from commercial fertilizer and manure applications.

Nitrogen/phosphorus pollution contributes significant loadings of nitrogen and phosphorus to waters of the United States and is one of the leading causes of water quality degradation. Many of our nation's waters, including rivers, canals, lakes, estuaries, and coastal marine waters, are affected by nitrogen/phosphorus pollution. There is increasing evidence of nitrogen/phosphorus pollution in Florida's waters and clear, widespread indications of the resulting adverse effects on aquatic life in those waters.

EPA is seeking to improve and enhance protection of aquatic life from the detrimental effects of nitrogen/phosphorus pollution through the derivation and implementation of numeric nutrient criteria. Numeric criteria and water quality standards are key components of water quality assessments and watershed protection management. These numeric criteria will create environmental baselines that allow Florida to manage waters more effectively, measure progress, and support broader partnerships based on nutrient trading, best management practices (BMPs), land stewardship, wetlands protection, voluntary collaboration, and urban stormwater runoff control strategies. Establishing numeric criteria in State water quality standards will give Florida greater ease and faster development of total maximum daily loads (TMDLs), provide quantitative targets to support trading programs, enable permit writers greater flexibility and ease in developing National Pollutant Discharge Elimination System (NPDES) permits, and increase the effectiveness in evaluating the success of Florida's efforts to control nitrogen/phosphorus pollution throughout the State.

Chapter 3 describes the approach EPA is considering to develop numeric criteria for Florida's estuaries. The approach described would allow the Agency to most fully consider characteristics of estuarine ecosystems (e.g., water quality and biological communities in estuaries are affected by a combination of basin shape, tides, and the magnitude, location, and quality of freshwater inflows). In some of Florida's estuaries, the semi-enclosed basins that define their spatial extent may also create sub-regions with differentiated water quality and aquatic life uses, which could also result in water quality criteria specific to a particular sub-region. The approach EPA is considering for deriving numeric nutrient criteria for estuarine waters in South Florida differs from that outlined in Chapter 3 because the water systems in South Florida are unique due to the high degree of management of the waters. Methods for South Florida are described in Chapter 5.

EPA's methodology first delineates the estuaries into discrete areas around Florida's coastline for the purpose of organizing the criteria development process. Each of these discrete areas will then be evaluated to determine the appropriate "assessment endpoints" and "measurement endpoints." The specific endpoints and indicators that EPA is considering for use in the development of numeric criteria in Florida's estuaries include protection and restoration of healthy seagrass communities, balanced communities of benthos, plankton, and nekton, and balanced algal biomass and production. Chapter 3 discusses the rationale that may be used for selecting specific water quality variables for each of the estuaries. Finally, EPA is considering three approaches: (1) reference conditions, (2) stressor-response relationships, and (3) water quality simulation modeling that could be used independently or in combination to develop numeric criteria for chlorophyll *a*, total nitrogen (TN) and total phosphorus (TP).

Chapter 4 provides the rationale and approaches EPA is considering to derive numeric criteria for Florida's coastal waters. For much of the Florida's coastal waters, EPA is considering a reference-based approach with satellite-derived chlorophyll *a* (Chl_{RS}-a) observations. Satellite ocean color remote sensing technology has advanced over the past decade and historical Chl_{RS}-a data are available for the past ten years. In contrast there is relatively little field monitoring data of chemical and biological constituents along the Northwest Gulf Coast and Atlantic Coast of Florida.

Coastal physical forcings such as wind, currents, and tides are known to influence coastal chlorophyll dynamics together with nutrient loadings from the land. Thus, all of these processes will be represented when using remote sensing as a reference condition approach. Specifically, EPA is considering the use of remote sensing data to develop numeric criteria for the Northwest Gulf Coast, West Gulf Coast, and Atlantic Coastal Areas of Florida. Due to interference from colored dissolved organic matter and bottom reflectance on satellite measurements, EPA is not considering the derivation of numeric criteria using remote sensing data in coastal waters from Apalachicola Bay to Suwannee River (Big Bend) and South Florida.

Chapter 5 outlines the approach EPA is considering to derive numeric criteria for nitrogen/phosphorus pollution in South Florida marine and inland flowing waters. EPA is defining South Florida inland flowing waters as free-flowing, predominantly fresh surface water in a defined channel, and includes, streams, rivers, creeks, branches, canals, freshwater sloughs, and other similar water bodies located in the South Florida nutrient watershed region. South Florida marine waters include estuarine and coastal waters extending three nautical miles offshore.

EPA is considering a reference-based approach to derive numeric TN and TP criteria for South Florida inland flowing waters and numeric chlorophyll *a*, TN, and TP criteria in South Florida marine waters using least-disturbed sites that support balanced natural populations of aquatic flora and fauna. Alternative methods of criteria derivation for inland flowing waters that EPA is considering include stressor-response relationships between chlorophyll *a* and TN and TP, and a distributional approach using all sites. EPA is not establishing new TP criteria for canals in the Everglades Protection Area (EvPA) in deference to the Everglades Forever Act (EFA).

Pursuant to 40 CFR 131.10(b), water quality standards must ensure the attainment and maintenance of downstream water quality standards. Thus, EPA is deriving numeric nutrient criteria for streams in Florida in order to protect the estuarine waterbodies that ultimately receive nitrogen/phosphorus pollution from the watershed. These criteria, which EPA will refer to as

Downstream Protection Values, or DPVs, will apply in place of the stream's TN and TP criteria if the applicable DPV is more stringent.

The conceptual approach that EPA is considering for developing stream DPV criteria will begin with estimates of limits on TN and TP loading rates that are needed to support balanced natural populations of aquatic flora and fauna in estuarine waters. The loading limits will be determined as part of the criteria development effort for estuarine waters as described in Chapter 3 of this document. The protective load limits can be scaled by average streamflow entering the estuary to determine criteria for TN and TP concentrations in streams as they discharge into estuaries. Finally, DPVs can be determined for upstream reaches within watersheds by accounting for expected loss or permanent retention of TN and TP within the stream network. Because of the complexities associated with the managed flows in South Florida inland flowing waters (Chapter 5), the fraction of TN or TP from the upstream tributary reach that eventually flows into marine waters cannot be estimated or predicted. Therefore, EPA is considering expressing DPVs at the terminal reach of the tributary into an estuary as protective concentrations or, alternatively, protective loads.

1 Introduction

1.1 Purpose of the Document

The United States Environmental Protection Agency (EPA) is deriving numeric criteria for nitrogen/phosphorus pollution to protect waters in the State of Florida, in response to the 2009 determination that "new or revised water quality standards for nutrients in the form of numeric nutrient criteria are necessary [in the State of Florida] to meet the requirements of Clean Water Act (CWA) (CWA section 303(c)(2)(A) and 40 CFR §131.11(a)(1))." This document describes approaches EPA is considering for the derivation of numeric criteria in Florida estuaries, coastal waters, and southern inland flowing waters. In addition, this document describes the approach EPA is considering for developing downstream protective values (DPVs) that may be applied to Florida streams to ensure the attainment and maintenance of downstream water quality standards.

1.2 Document Organization

Chapter 1 provides a summary of the background information that serves as a basis for the development of numeric criteria for controlling nitrogen/ phosphorus pollution. Chapter 2 includes the statutory requirements that direct EPA and States to reduce nitrogen/phosphorus pollution, the regulatory action that determined numeric criteria are necessary in the State of Florida, background information on nitrogen/phosphorus pollution and effects within the aquatic environment.

Chapter 2 provides a description of EPA's general approach for developing numeric criteria, generic conceptual model, and endpoints that form the basis for analysis plans.

The next three chapters summarize background information on the geography, water quality characteristics, and water quality conditions specific to each water body type and presents approaches EPA is considering for numeric criteria derivation. Potential uncertainties and data gaps associated with the approaches are also included.

Chapter 3 describes the approach EPA is considering for numeric criteria derivation in Florida estuaries.

Chapter 4 provides a description of the approach EPA is considering for numeric criteria derivation in Florida coastal waters.

Chapter 5 describes the approach EPA is considering for numeric criteria derivation to address the unique hydrodynamic conditions found in South Florida.

Chapter 6 describes an approach EPA is considering to quantitatively derive numeric values for streams to protect downstream estuaries. Potential uncertainties and data gaps associated with the approach are also included.

1.3 Background: Clean Water Act Requirements and Florida's Current Water Quality Standard for Nitrogen/Phosphorus Pollution

The Clean Water Act (CWA) established a basis for water quality protection in section 101(a): "The objective of this Act is to restore and maintain the chemical, physical, and biological integrity of the Nation's waters." Under CWA section 101(a)(2) "it is the national goal that wherever attainable, an interim goal of water quality which provides for the protection and propagation of fish, shellfish, and wildlife and provides for recreation in and on the water be achieved." This goal is commonly referred to as fishable and swimmable.

In order to meet the fishable and swimmable goal, the CWA defines a structure of interlinked programs and identifies the establishment of water quality standards as the key component necessary to achieve that goal. In many ways, water quality standards provide the common mechanism by which the other parts of the CWA (such as NPDES permits and TMDLs) work together to accomplish the overall goals and objective of the CWA.

To assist in achieving the goals and objective of the CWA, States adopt water quality standards that consist of designated uses of the navigable waters and water quality criteria that are protective of those designated uses. The State specifies the designated use that must be achieved or protected. When designating uses, the State must take into consideration the use and value of water for public water supplies, protection and propagation of fish, shellfish and wildlife, recreation in and on the water, agricultural, industrial and other purposes including navigation (CWA section 303(c) and 40 CFR 131.10).

Designated uses can be general, such as aquatic life use protection or primary contact recreation, or more specific. In Florida, waters have been already classified by designated use.

As described by FDEP in the *2010 Integrated Water Quality Assessment for Florida* (FDEP 2010):

> Florida's Water Quality Standards Program, the foundation of the state's program of water quality management, designates the "present and future most beneficial uses" of the waters of the state (Subsection 403.061[10], F.S.). Florida's surface water is protected for five designated use classifications, as follows:
>
> - Class I Potable water supplies
> - Class II Shellfish propagation or harvesting
> - Class III Recreation, propagation, and maintenance of a healthy, well-balanced population of fish and wildlife
> - Class IV Agricultural water supplies (large agricultural lands, located mainly around Lake Okeechobee)
> - Class V Navigation, utility, and industrial use (there are no state waters currently in this class)

> Class I waters generally have the most stringent water quality criteria and Class V the least. Class I, II, and III surface waters share water quality criteria established to protect recreation and the propagation and maintenance of a healthy, well-balanced population of fish and wildlife. All waters of the state are considered to be Class III, except for those specifically identified in Section 62-302.600, F.A.C. All waters of the state are required to meet the "Minimum Criteria for Surface Waters," as identified in Section 62-302.500, F.A.C.

In general, States adopt water quality criteria into water quality standards to protect the designated uses from the discharge of pollutants. These criteria are expressed as either narrative statements or numeric values. In order to protect the designated uses listed above from nitrogen/phosphorus pollution, the State of Florida has adopted a narrative criterion into their water quality standards, which provides in part:

> 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 (F.A.C. 62-302-530(47)(b)).

On January 14, 2009, EPA made a determination under CWA section 303(c)(4)(B) that numeric nutrient water quality criteria for lakes, flowing waters, estuaries and coastal waters are necessary for the State of Florida to meet the requirements of CWA Section 303(c). Upon making this determination, the EPA Administrator is then required to promptly prepare and publish proposed regulations setting forth new or revised water quality standards.

In making the determination, the Agency considered (1) the State's documented unique and threatened ecosystems, (2) the high number of impaired waters due to existing nitrogen/phosphorus pollution, and (3) the challenge associated with growing nitrogen/phosphorus pollution resulting from expanding urbanization, continued agricultural development, and a significantly increasing population. EPA also reviewed the State's regulatory nutrient accountability system, which represents an impressive synthesis of technology-based standards, point source control authority, and authority to establish enforceable controls for nonpoint source activities. However, the significant challenge faced by the water quality components of this system is its dependence upon an approach involving resource-intensive and time-consuming site-specific data collection and analysis to interpret the narrative nutrient criterion. EPA subsequently determined that Florida's reliance on a case-by-case interpretation of its narrative nutrient criterion in implementing an otherwise comprehensive water quality framework of enforceable accountability was insufficient to ensure protection of applicable designated uses.

EPA determined that numeric nutrient criteria would strengthen and expedite the process for identifying impaired waters, preparing TMDLs and deriving water quality-based effluent limits in NPDES permits, thus providing the necessary protection for the State's designated uses. In November 2010, EPA established criteria for Florida's lakes, flowing waters[1] and springs within

[1] In the November 2010 rulemaking, EPA did not establish numeric criteria for inland flowing waters in South Florida. For the purpose of this effort, EPA has distinguished South Florida as those areas south of Lake Okeechobee and the Caloosahatchee River watershed to the west of Lake Okeechobee and the St. Lucie watershed

the State of Florida. These promulgated criteria only apply to predominantly fresh surface waters classified as Class I or Class III in order to implement the State's narrative nutrient criterion mentioned above. EPA will propose numeric criteria for nitrogen/phosphorus pollution to protect estuaries, coastal areas and South Florida inland flowing waters that have been designated Class I, II and III. The methods and approaches that EPA is considering to establish numeric criteria for estuaries, coastal areas and South Florida inland flowing waters is the subject of this document.

1.4 Nature of the Chemical Stressor: Nitrogen/Phosphorus Pollution

Excess anthropogenic inputs of nitrogen and phosphorus, commonly referred to as nutrient pollution or nitrogen/phosphorus pollution, in surface waters can result in excessive and imbalanced primary production in a waterbody, referred to as eutrophication.[2]

1.4.1 Stressor Source and Distribution

Nitrogen/phosphorus pollution in water bodies comes from many point and nonpoint sources, which can be grouped into the following five major categories: 1) urban stormwater runoff, 2) municipal and industrial waste water discharge (e.g. sewage effluent, landfill leachate), 3) row crop agriculture (e.g. commercial fertilizer and manure applications), 4) animal husbandry, and 5) atmospheric deposition (and fossil fuel combustion) (SENITG 2009). These sources are often direct inputs to estuaries and coasts because of the large populations that reside very close to their shores. Estuaries and coastal waters are especially vulnerable to nitrogen/phosphorus pollution because they receive nitrogen and phosphorus from multiple natural and anthropogenic upstream sources.[3]

1.4.2 Aquatic Ecosystem Effects of Nitrogen/Phosphorus Pollution

The biennially published National Water Quality Inventory Report to Congress indicates that excess nitrogen and phosphorus are consistently a major source of water quality impairment in the Nation's waters. Since the 1992 report, nitrogen and phosphorus compounds have consistently ranked in the top five causes of U.S. water quality impairment. This pollution causes major impacts to aquatic ecosystems and disrupts the natural populations of flora and fauna (Dodds et al. 2009; Howarth et al. 2002; National Research Council 2000). Imbalances in natural communities can adversely affect aquatic life as well as human health. In Florida, nutrients have been reported as the cause of impairment in 120 estuarine water bodies covering approximately 569 square miles (FDEP 2010).

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to the east of Lake Okeechobee. Methods and approaches that EPA is considering to establish criteria for the inland flowing waters in South Florida are contained within this document.

[2] Eutrophication is defined as an increase in the rate of supply of organic matter to an ecosystem (Nixon 1995). Eutrophication can adversely affect human and aquatic life uses of waters, and may impact human health. Here we are concerned with eutrophication caused by excess loading of N and P, which causes increased organic matter production within the ecosystem.

[3] Some estuaries can be more susceptible to eutrophication than others. Physiographic setting, primary production base, nutrient load, dilution, residence time, flushing, stratification, hypsography, grazing of phytoplankton, suspended material load and light extinction, denitrification, spatial and temporal distribution of nutrient inputs, and allochthonous organic matter inputs all affect an estuary's response to added nutrients (National Research Council 2000).

The impacts of nitrogen/phosphorus pollution can adversely affect aquatic life in many different ways (see Figure 1-1). The effects of nitrogen/phosphorus pollution include direct changes to aquatic systems (e.g., increased algal growth, changes in algal species composition, and increased organic matter production) and indirect effects (e.g., loss of submerged aquatic vegetation, nuisance algal blooms, and low dissolved oxygen) (USEPA 2006, 2008a). The eutrophication process has resulted in large "dead zones" found in many coastal areas, such as the Gulf of Mexico and Chesapeake Bay (Ecological Society of America 2009) and reduced seagrass beds, a foundation species for many estuarine waters (Hughes et al. 2009; Tomasko et al. 2005). The Florida Department of Environmental Protection (FDEP) notes that harmful algal blooms in both fresh and marine waters continue to be a concern that have toxic effects to fish, wildlife, and humans, and indirect effects due to anoxia (FDEP 2010). In 2008 and 2010, FDEP lists nutrient impairment as a special initiative (FDEP 2008, 2010).

Environmental consequences from changes in primary production (e.g., increases in phytoplankton) can include increased turbidity and decreased light penetration (Boyer et al. 2009; Bricker et al. 2007; Bricker et al. 2008; McPherson and Miller 1994). This can reduce light availability necessary for the growth of submerged aquatic vegetation. This effect often leads to declines in seagrass in estuaries and coastal waters (Lee et al. 2007; Dennison 1987; Duarte 1991). Seagrass are critical components of many estuarine and coastal systems and are used as feeding, spawning, and nursery grounds for many aquatic species (see review by Waycott et al. 2009).

Imbalances in primary producer dynamics can cause changes in habitat and available food resources that can induce changes affecting an entire food web (Bricker et al. 2003b; Vitousek et al. 1997). Increased phytoplankton abundance has also been linked to composition shifts to less desirable species (Paerl 1988). Because these changes affect natural processes at the lowest levels of the ecosystem, they can cause a cascade of problems.

Eutrophication has also been shown to increase the incidence of disease in aquatic animals and wildlife (Johnson et al. 2010). Although nitrogen/phosphorus pollution may not always be the trigger, nutrient overenrichment can contribute to blooms of nuisance or toxic algae (Glibert et al. 2006) or may extend bloom duration (Vargo 2009b). Called harmful algal blooms (HABs), these blooms can damage or clog the gills of fish and invertebrates and cause illness or death to animals and humans (Falconer 1999; NOAA 2010). Direct impacts to humans result from exposure to HAB toxins or consumption of toxic shellfish. Examples of freshwater algal species that are considered HABs in Florida include *Microcyctis*, *Arabaena*, and *Cylindrospermopsis* and marine species include *Karenia brevis*, *Alexandrium monilatum*, *Takayama pulvhella*, *K. selliforms*, *K. mikimotoi*, *Karlodinium venificum*, *Pyrmnesium parvum*, and *Chattonella spp*. (FDEP 2010).

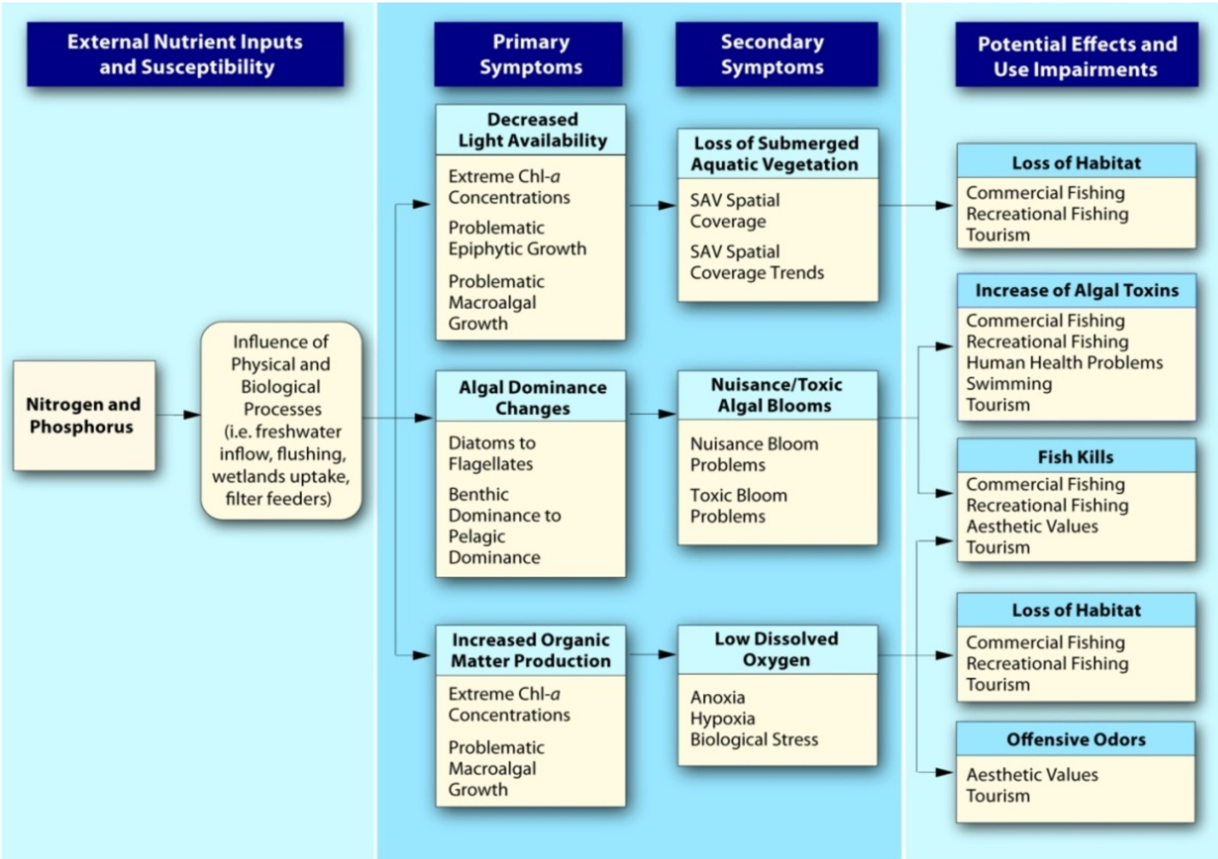

Figure 1-1. The model of primary and secondary symptoms of nitrogen/phosphorus pollution and the potential effects and impairments (Bricker et al. 1999; Bricker et al. 2003b)

Excessive algal growth contributes to increased oxygen consumption associated with decomposition, potentially reducing oxygen to levels below those needed for aquatic life to survive (NOAA 2010; USGS 2010). Low oxygen concentrations, or hypoxia, can occur in episodic "events," which sometimes develop overnight. Migration to avoid hypoxia depends on species' mobility, availability of suitable habitat, and adequate environmental cues for migration. For example, mobile species, such as adult fish, can sometimes survive by moving to areas with more oxygen availability. Less mobile or immobile species, such as oysters and mussels, cannot move to avoid low oxygen and are often killed during hypoxic events (Ecological Society of America 2009). While certain mature aquatic animals can tolerate a range of dissolved oxygen (DO) levels, younger life stages of species like fish and shellfish often require higher levels of oxygen to survive (USEPA 2000a). Sustained low levels of DO cause a severe decrease in the amount of aquatic life in hypoxic zones and affect the ability of aquatic organisms to find necessary food and habitat. In extreme cases, anoxic conditions occur when and where there is a complete lack of oxygen. Because most plants and aquatic organisms cannot live without sufficient oxygen, hypoxic and anoxic areas are sometimes referred to as dead zones (Ecological Society of America 2009).

Other indirect impacts of algal blooms include restrictions on recreation (such as boating, swimming, and kayaking) due to closures of areas to recreational uses. The loss of biological resources can also reduce or preclude recreational fishing, shellfish harvest, and diving. Other direct impacts to humans include harmful levels of nitrate in drinking water supplies (SENITG 2009).

1.5 Purpose of this Effort

As described above, EPA will propose numeric values to translate the State of Florida's current narrative criteria to protect estuaries, coastal waters and South Florida inland flowing waters from nitrogen/phosphorus pollution. These numeric criteria will establish limits to pollutant concentration levels that will ensure protection of the designated use of the water body that has been determined by the State. Specifically, the numeric values will protect estuaries, coastal waters and South Florida inland flowing waters within the State that have been designated as Class I, II or III (see Section 1.3), translating the currently applicable narrative criterion that the State has established to be protective of these designated uses:

> 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 (F.A.C. 62-302-530(47)(b))

EPA is not considering the establishment of numeric values to protect Class IV and V waters as they are not part of the CWA section 101(a) designations that require water quality that provide for fishable and swimmable conditions. As the State of Florida already has numeric criteria for dissolved oxygen, EPA is not considering the development of dissolved oxygen criteria. For those Class I waters in South Florida (i.e., designated as potable water supplies), the drinking water maximum contaminant levels for nitrates (10 mg/L) would continue to apply. EPA is currently not considering the modification of any designated uses of waters within the State of Florida.

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2 Approach

2.1 Overview

EPA's 1976 publication entitled *Quality Criteria for Water* (also known as the Red Book) contains ambient water quality criteria for nitrates and elemental phosphorus. For domestic water supplies, the maximum contaminant level for nitrate was set at 10 mg/L to protect human health from exposure to this pollutant through domestic drinking water. The phosphorus criterion was set at 0.10 µg/L elemental phosphorus to protect against the toxic effects of elemental phosphorus to estuarine and marine organisms. Note that neither of these criteria was set to reduce the potential for eutrophication, although the Red Book does present a rationale for supporting a total phosphorus criterion.

EPA has published peer-reviewed technical guidance for states to develop numeric nutrient criteria for lakes and reservoirs (USEPA 2000b), for rivers and streams (USEPA 2000c), for estuarine and coastal waters (USEPA 2001), and for wetlands (USEPA 2008b). These guidance manuals are intended to help states, tribes and others in establishing scientifically defensible nutrient criteria for classes of water bodies. EPA has also published supplemental peer reviewed technical guidance for states using stressor-response relationships to derive numeric nutrient criteria (USEPA 2010).

Additionally, EPA has recommended CWA section 304(a) water quality criteria for nutrients with the aim of reducing and preventing eutrophication on a national scale. There are a total of 26 peer-reviewed ecoregional criteria documents in 2001 and 2002 that cover most water body types in the United States (12 lakes and reservoirs, 13 rivers and streams, and one wetland). Each criteria document presents recommended criteria for causal parameters (total phosphorus and total nitrogen) and response variables (chlorophyll *a* and some form of water clarity, i.e., turbidity or Secchi depth). EPA developed the ecoregional criteria values using a distributional approach that utilized all available data. This information is intended as a starting point for states, authorized tribes and others to develop more refined numeric criteria, as appropriate, using EPA waterbody-specific technical guidance manuals and other scientifically defensible approaches. These recommended criteria documents can be accessed at the following Web sites:

- Lakes: http://www.epa.gov/waterscience/criteria/nutrient/ecoregions/lakes/index.html
- Streams and Rivers: http://www.epa.gov/waterscience/criteria/nutrient/ecoregions/rivers/index.html
- Wetlands: http://www.epa.gov/waterscience/criteria/nutrient/ecoregions/wetlands/wetlands_13.pdf

None of these available recommended numeric criteria apply to Florida's estuaries, coastal waters and southern inland flowing waters. Thus, for this effort EPA must derive numeric values to translate Florida's existing narrative criterion using the published peer reviewed technical guidance, the best available data, and sound scientific rationale.

2.1.1 Nutrient Criteria Development Guidance

As noted above, EPA published peer reviewed technical guidance for developing numeric nutrient criteria for rivers and streams in July 2000 (USEPA 2000c), and estuaries and coastal marine waters in October 2001 (USEPA 2001). These technical guidance documents describe the techniques used to derive numeric criteria for use in State water quality standards. They provide background information on classifying water bodies, selecting criteria variables, designing monitoring programs, analyzing nutrient and algal data, deriving regional criteria, and implementing management practices.[4]

The documents describe three general approaches that could be used to develop numeric nutrient criteria (USEPA 2000c):

1. Identification of reference conditions for each water body type based on best professional judgment or percentile selections of data plotted as frequency distributions
2. Use of predictive relationships (e.g., trophic state classifications, empirical and mechanistic models, biocriteria)
3. Application and/or modification of established nutrient/algal thresholds (e.g., nutrient concentration thresholds or algal limits from published literature)

EPA's technical guidance documents suggest that each of the above analytical approaches was appropriate for deriving scientifically defensible numeric nutrient criteria. However, EPA recognized that each approach has different data requirements, and these differences should be considered in the context of individual situations and available information. The methods and approaches described in this document demonstrate that EPA is considering the derivation of numeric criteria for estuarine, coastal, and South Florida inland flowing waters using the tools and approaches described in these guidance documents, as well as new methods that follow the general approach outlined by EPA and are reflective of the latest scientific knowledge.

2.1.2 General Approach

The general approach that EPA followed for each of the water body system types is outlined in the estuarine and coastal marine waters guidance document (USEPA 2001):

1. **Establish a panel of technical experts** – this group is responsible for developing the numeric criteria; EPA created an internal work group consisting of staff from EPA's Office of Science and Technology, Office of Research and Development (staff from the Gulf Ecology Division in Pensacola FL and the Atlantic Ecology Division in Narragansett, RI), and EPA Region 4.

2. **Review the scientific and regulatory basis** – Chapter 1 of this document describes the scientific and regulatory basis for establishing numeric criteria for Florida estuarine, coastal, and South Florida waters. In addition, Chapter 1 describes the basis for EPA to establish numeric downstream protective values to protect estuaries from impacts originating upstream.

[4] This and other EPA guidance documents can be accessed from the Office of Science and Technology's Web site: http://www.epa.gov/waterscience/criteria/nutrient/guidance/index.html.

3. **Develop a classification scheme** – this step establishes a classification scheme for subdividing the population of waterbodies for which numeric criteria are developed. For this effort, EPA first divided the waters into three groups: (1) estuarine, (2) coastal, and (3) South Florida. Chapters 3-5 provide additional details on the specific classifications for each group of Florida waters.
4. **Select indicator variables** – EPA has evaluated causal (e.g., TN and TP) and response (e.g., chlorophyll *a*, and others) specific to the classifications of waters in the groups of estuarine (see Chapter 3), coastal (see Chapter 4), and South Florida (see Chapter 5). Based on the evaluations and analyses, EPA selected appropriate causal and response variables for each water body system in the development of numeric criteria.
5. **Data collection and assessment** – based on the classification scheme, EPA will collect available data for each system or group of waters. EPA has reviewed data from STORET, Florida's IWR data set, and NOAA and other remote sensing data.
6. **Establish methodology** – based on the availability of data and assessment endpoints used to translate Florida's narrative criterion, EPA is considering a variety of specific methods for deriving the numeric criteria. Chapters 3-5 describe the methodologies in detail.
7. **Criteria development** - the EPA guidance *Nutrient Criteria Technical Guidance Manual: Estuarine and Coastal Marine Waters* (USEPA 2001) outlines the following process for developing numeric nutrient criteria:
   - Examination of the historical record or paleoecological evidence for evidence of a trend.
   - Determination of a reference condition using one of several alternative approaches. Remember that the reference condition, however derived, is only one of the three approaches of the criteria development process.
   - Use of empirical modeling (or surrogate data sets, where available, in those instances where insufficient information exists). This may be the case especially in estuaries with insufficient hydrological data, or significantly developed or modified watersheds.
   - Objective and comprehensive interpretation of all of this information by the established panel of technical experts.
   - Finally, the criterion developed for each variable should reflect the nutrient condition for the waterbody to protect the designated use. Second, it must be reviewed to ensure that the proposed level does not entail adverse nutrient loadings to downstream waterbodies.

   EPA is considering this criteria development process for each of the groups of water bodies and explains the process in more detail for estuarine waters (see Chapter 3), coastal waters (see Chapter 4), and South Florida marine and inland flowing waters (see Chapter 5).

2.2 Conceptual Model

Ideally, the aquatic life to be protected in a water body needs to be characterized in a way that captures the range and diversity of the inter-related life forms that comprise the biological community that the public expects to be protected. For this effort, EPA has considered the available data and information, which has provided insight into the relative health of various systems in Florida, all potentially subject to degrees of stress from various levels of human disturbance. In order to restore and maintain water quality, it is necessary to determine the health of the system, and to understand the range of conditions, both physical and chemical, that support and sustain that health. To accomplish this, one can select suitable surrogates or indicators closely correlated with overall system health, and expected to be sensitive to stressors, in this case, nitrogen/phosphorus pollution. Figure 2-1 illustrates the conceptual relationship between the objective, which is the support of balanced natural populations of aquatic flora and fauna, appropriate *biological assessment endpoints* and *indicators* (the causal and response variables, or measurement endpoints, for numeric criteria). EPA also provides a simple flow chart analysis plan for each group of waters in Chapters 3-5.

Nitrogen/phosphorus pollution can result in excess biomass which can deplete oxygen resulting from decay of nutrient-enhanced organic matter production. Recognition of this pathway provides direction for setting protective numeric criteria for nitrogen/phosphorus and related parameters. In order to support balanced natural populations of aquatic flora and fauna in Florida through effective implementation of CWA programs, EPA will develop and establish numeric criteria for causal variables, TN and TP, as well as the primary response variable, chlorophyll *a*.

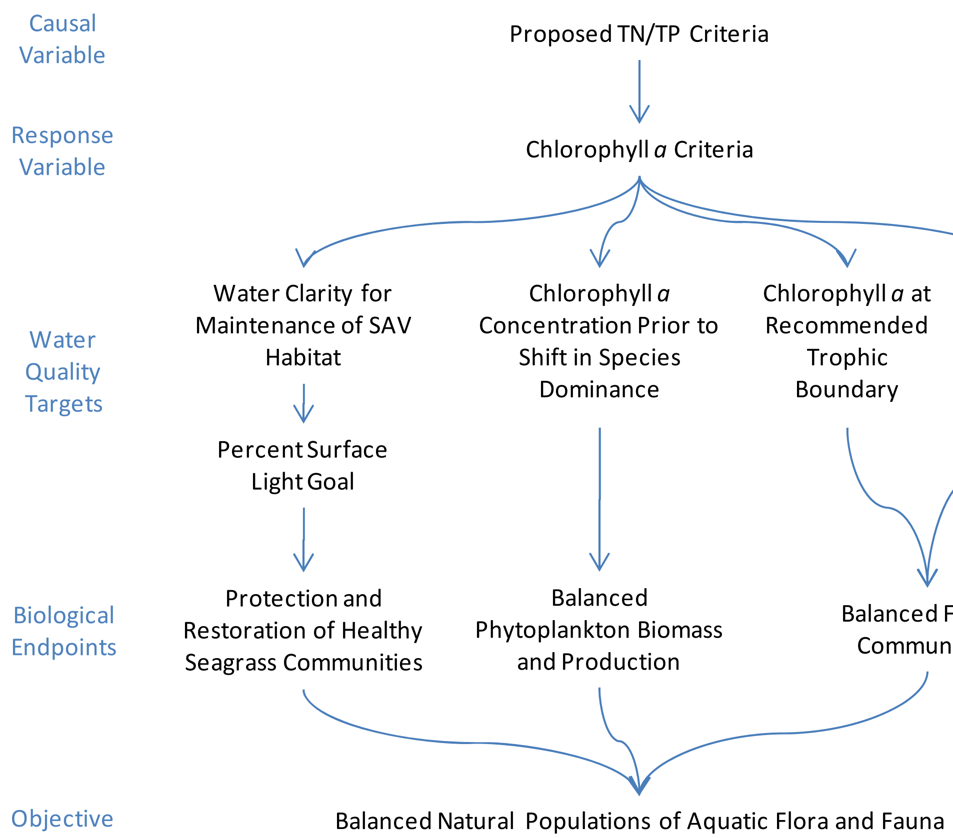

Figure 2-1. Pathways for nutrient effects on estuarine and coastal aquatic life uses.

2.3 Review of Scientific Basis and Selection of Potential Assessment Endpoints and Indicator Variables

EPA conducted a thorough literature review to evaluate biological, chemical, and physical assessment endpoints appropriate to the protection of aquatic flora and fauna from nitrogen/phosphorus pollution (see Appendix B).

The true assessment endpoints are the valued ecosystem characteristics that are desired to be protected. In a regulatory context, the designated uses and their associated narrative criteria may be considered as assessment endpoints. These assessment endpoints (such as shellfish propagation and harvesting) are often difficult to predict or measure directly. Therefore, the development of water quality criteria usually proceeds through the evaluation of simpler endpoints (referred to as indicators or measures) that are measurable and predictable, and serve as surrogate measures to link stressors and outcomes.

These "measures" include measures of effect (formerly known as "measurement endpoints"), defined as "measurable changes in an attribute of an assessment endpoint or its surrogate in response to a stressor to which it is exposed," measures of exposure, defined as "measures of stressor existence and movement in the environment and their contact or co-occurrence with the assessment endpoint," and measures of ecosystem and receptor characteristics (USEPA 1998). The TMDL and Watershed Approach literature tends to refer to these measures as "indicators."

A target is simply a value of an indicator that is consistent with attaining the assessment endpoint or management objective. In other words, a target is equivalent to a criterion value for protecting a specific use at a given site. For this effort, the target will be the numeric value of an indicator variable that supports a balance natural population of aquatic flora and fauna in Florida's estuaries, coastal areas and southern inland flowing waters.

Salient aspects of the literature review and EPA's basis for selecting assessment endpoints and the proposed water quality indicator variables to protect those endpoints are discussed in subsequent sections of this chapter.

2.3.1 Selecting Assessment Endpoints and Water Quality Indicator Variables

Selecting assessment endpoints to protect a balanced natural population of aquatic flora and fauna represents a balance among environmental sensitivity to nitrogen/phosphorus pollution and available data. To develop numeric criteria, it is important to select assessment endpoints that are sensitive to nitrogen/phosphorus pollution, so that one can infer that the numeric criteria will protect less sensitive receptors from such pollution. Additionally, it is important to choose endpoints with sufficient data that would allow quantitative relationships to be developed either through stressor-response relationships (e.g., empirical or regression models) and/or water quality simulation models, and that would be sensitive to environmental changes that are supported by data.

There are numerous endpoints that can, at a minimum, be qualitatively related to nutrient enrichment (e.g., Bricker et al. 2008). EPA searched scientific databases (including: Google

Scholar, Web of Science, and state research and agency reports) and reviewed more than 800 documents to investigate assessment endpoints and the likely stressors driving responses in estuarine and marine systems. The assessment endpoints examined include phytoplankton, macroalgae, epiphytes, seagrass, benthic macroinvertebrate and fish indices, HABs, and coral. The literature review also captured the nature and location of the investigations and the endpoint's relationship to nutrients and other causal variables. A detailed bibliography and results of the literature review are provided in Appendix A and Appendix B, respectively. The major assessment endpoints and indicator variables considered and salient aspects of the literature review are summarized in Table 2-1 and Table 2-2, respectively. For a discussion on assessment endpoints considered in the development of numeric criteria for South Florida inland flowing waters, please see Chapter 5.

Table 2-1. Assessment endpoints for evaluating the magnitude and effects of nutrients, including advantages and disadvantages.

	Importance	Linkages to, or Effects of, Nutrients	Advantages	Disadvantages
Seagrass	• Valuable marine habitat • Primary food source for many organisms	• Spatial extent, density, growth rates decline with decreased light transmittance • Light transmittance decreases with decreased clarity/increased nutrients • Light requirement usually 20–25% surface irradiance	• Mechanism of nutrient impact mostly well-understood • Colonization depth (Z_c) useful indicator • Once Z_c goal established, can use light requirements to infer water clarity requirement and chlorophyll a criteria • Historical depth of colonization could be used to infer reference water clarity	• Co-factors exist – salinity stress, food web change, dredging, propeller scarring, sediment loading, disease • Response to nutrients can be slow (especially recovery)
Phytoplankton	• Primary producers and important component of marine food web • Excess growth affects clarity, DO, habitat, aesthetics	• Nutrients are key limiting factors for algal growth rate.	• Responsive to nutrients, Well-established basis for use as indicator • Biomass data in estuarine waters are routinely monitored and data are generally abundant • Satellite-derived chlorophyll data readily available in many coastal waters	• Other factors can interfere with evaluating stressor-response relationships • Species composition data limited; differences in field sample and taxonomic methods may increase uncertainty • Field-collected biomass data in coastal (offshore) waters are limited • Most estuaries lack species composition models developed for nutrient response • Lack of phytoplankton data in healthy canals

	Importance	Linkages to, or Effects of, Nutrients	Advantages	Disadvantages
Harmful Algal Blooms	• Certain HABs impact human health/other marine organisms, and aquatic ecosystems • Often associated with toxins leading to faunal kills, shellfish contamination, economic impacts, decline in aesthetic value, environmental and ecological damage	• HAB species other than K. *brevis* occur in Florida marine waters, but are less studied	• Foul odor and reduced aesthetics can lead to public awareness • Once driven toward landfall, there is some evidence that *K. brevis* bloom duration may be extended by land-based sources of nutrients	• *K. brevis* initiation occurs in coastal waters beyond 3 miles • It is unclear that reduction in land-based nutrients would reduce *K. brevis* blooms. Current Gulf of Mexico *K. brevis* models (see Appendix C) are focused on research applications
Corals	• Highly productive and valued ecosystem • High species richness and diversity	• Nutrient-poor habitat • Nutrients may contribute to bleaching, disease, and excess macroalgal growth	• Highly valued resource	• Role of nutrients on coral health is mixed • Method limitations • Interacting factors are important (dissolved organic carbon, fish, etc.) • May depend on duration of enrichment
Epiphytes	• Excess growth hinders seagrass growth	• Epiphyte biomass increases with nutrient enrichment	• Responsive to nutrients • May be more sensitive than seagrass loss, especially epiphyte composition • Clear linkage to important aquatic life (seagrass)	• Biomass responses sometimes equivocal • Confounding factors (light, grazing, etc.) • Composition difficult to measure • Limited data
Invertebrates	• Reliable indicator of biological conditions	• Invertebrate community changes from increased phytoplankton food base and reduced benthic food base • Severe community changes with hypoxia	• Established indicator of biological conditions • Existing monitoring programs • Stream Classification Index in canals decreases with increasing nutrient concentration	• Many confounding factors (e.g., seagrass and other habitat loss, sediment toxicity, overfishing, indirect effects of nutrients)
Fish	• Indicator of biological condition	• Nutrient loading may impact habitat quality for fish (e.g., due to hypoxia or seagrass loss) HABs can cause fish mortality or reduced fish growth. • Excess nutrients can also stimulate fisheries production by increasing prey abundance. •	• Highly visible • Substantial public concern	• Many confounding factors (e.g., overfishing, stocking, seagrass and other habitat loss, indirect effects of nutrients)

Table 2-2. Indicator variables for evaluating the magnitude and effects of nutrients, including advantages and disadvantages.

	Importance	Linkages to, or Effects of, Nutrients	Advantages	Disadvantages
Clarity	• Affects growth of plants and phytoplankton	• Nutrient enrichment enhances phytoplankton growth, reducing clarity	• Easy to measure (photosynthetically active radiation [PAR], Secchi) • Clear linkage to important aquatic life (e.g., seagrass) • Sensitive to nutrient enrichment • Responsive to water quality management	• Confounding factors (e.g., inorganic particles, dissolved organic carbon [DOC])
Dissolved Oxygen	• Hypoxia kills fish and invertebrates • Hypoxic or low DO areas nullified as suitable habitat	• Nutrients affect organic loading through algal growth, depleting oxygen • Nutrients accelerate decomposition rates by microbial stimulation, consuming oxygen	• Existing criteria • Well established basis for protection of aquatic life • Clear linkages to nutrient enrichment • Extensive database	• Need to model relationship between nutrients and DO
Chlorophyll a	• Chlorophyll is an indicator of phytoplankton production and biomass	• Nutrients are key limiting factors for algal growth	• Responsive to nutrients • Biomass is a well-established as indicator of phytoplankton production • Biomass data in estuarine waters are routinely monitored and data are generally abundant • Satellite-derived chlorophyll data readily available in many coastal waters	• Establishing protective concentrations for non-seagrass uses is less well studied • Other factors can interfere with evaluating stressor-response relationships • Field-collected biomass data in coastal (offshore) waters are limited • Lack of phytoplankton data in healthy canals
Total Nitrogen	• N is typically more limiting of algal growth than P in estuarine systems	• N directly related to phytoplankton production in N-limited systems	• Estuarine water quality best predicted in the short term by antecedent TN loading rates or freshwater discharge • TN concentration is associated with TN loading over the long term	• Nutrient transport and transformation processes complex
Total Phosphorus	• Algal production can be P-limited in areas with less soil P such as in South Florida	• P directly related to phytoplankton production in P-limited systems • P-limitation more common in spring when N loading is highest	• TP loading best predicts water quality response in P-limited systems • TP concentration is associated with influent TP loading over the long term	• Water quality response relationship less strong in N-limited systems

2.3.2 Selected Biological Assessment Endpoints

Based on the reviewed literature on potential assessment endpoints for nitrogen/phosphorus pollution in estuarine and marine systems, EPA is considering the following water quality goals to be achieved when translating Florida's narrative nutrient criterion into numeric values: (1) healthy seagrass communities, (2) balanced phytoplankton biomass and production, and (3) balanced faunal communities.

2.3.2.1 Healthy Seagrass Communities

Seagrass can be considered foundation species in aquatic systems, meaning that they provide the complex physical substrate necessary to support normal growth and reproduction of a wide range of species (Hughes et al. 2009). Seagrass provide habitat for many fish, birds, and invertebrates, and are also an important food source for endangered species such as manatees and green turtles. Without seagrass, entire habitats and their associated biotic communities can be replaced by less desirable and simplified assemblages that are less ecologically productive and have reduced taxonomic and functional diversity. Although seagrass are widely distributed in Florida, they are not present everywhere (Sargent et al. 1995).

Healthy seagrass communities depend on a variety of physical and biological factors to thrive. Among these, water clarity (i.e., light availability) is important and related to nutrient enrichment (e.g., Dennison 1987; Duarte 1991; Duarte 1995; Lee et al. 2007). Light is a critical variable for protecting and restoring seagrass. Water clarity can be negatively affected by an over-abundance of phytoplankton because the phytoplankton itself and the detritus associated with phytoplankton production absorb and scatter light before it reaches the seagrass. Increases in phytoplankton abundance as a result of nutrient enrichment cause increases in light attenuation, which can be important to seagrass communities as in estuaries such as Lemon Bay, Florida, where light attenuation due to phytoplankton is estimated to range from 12 to 39 percent (Tomasko et al. 2001). Other factors (e.g., color, suspended sediments) in addition to phytoplankton can also produce light attenuation. Seagrass water clarity requirements can be computed as the percent of surface irradiance required at a selected seagrass bed depth. Numeric criteria for chlorophyll *a*, the indicator variable, can be computed using the relationship between water clarity and chlorophyll *a*. Subsequently, numeric criteria for TN and TP can be computed using the relationship between chlorophyll *a* and TN or TP.

2.3.2.2 Balanced Phytoplankton Biomass and Production

In most of Florida's waters, healthy biotic communities depend on normal, balanced levels of phytoplankton abundance (Bricker et al. 1999; Bricker et al. 2003b). Chlorophyll *a* concentration is the measurement endpoint most often used to indicate balanced phytoplankton biomass and production (Boyer et al. 2009; Hagy et al. 2008). Several water quality management organizations (e.g., Tampa Bay Estuary Program, Charlotte Harbor National Estuary Program) within the state of Florida have already begun developing chlorophyll *a* targets for estuaries based on water clarity or DO goals. FDEP has also established a chlorophyll *a* monitoring threshold in their Impaired Waters Rule (IWR) of 11 μg/L for all estuaries of the State (F.A.C. 62-303.353); this value represents conditions above which a water body is identified as impaired, however waters that are below IWR thresholds are not considered "in attainment" of the narrative criterion (Giattina 2008). The natural dynamics of chlorophyll *a* vary among and occasionally within aquatic systems. To determine the appropriate water quality criteria, a variety of factors influencing the response of chlorophyll *a*, the assessment endpoint, should be considered.

Shifts in the composition of phytoplankton and zooplankton assemblages have been observed in estuarine and freshwater ecosystems in which nutrient loading rates are increased, both in Florida and elsewhere in the country (Arhonditsis et al. 2007; Armitage and Fong 2004; Cloern 2001; Cloern 1996). Although informative studies of phytoplankton assemblage composition are relatively uncommon, unusually high phytoplankton biomass (and thus, chlorophyll *a*) has been associated with proliferation of toxic or otherwise harmful species (Cloern 2001). One reason is that such species are not effectively controlled by planktonic grazers, allowing their biomass to increase. Species shifts may involve an increase in the abundance of unpalatable, toxic, or otherwise nuisance species that disrupt grazing and may negatively impact the estuarine food chain from the bottom up. Some species shifts occur in response to a change in the relative abundance of different nutrients. Increased abundance of nitrogen and phosphorus sometimes result in silica limitation, which favors non-diatom species because they do not require silica (Cloern 2001). EPA is considering using chlorophyll *a* as an indicator of the changes in phytoplankton species composition. Subsequently, numeric criteria for TN and TP can be computed using the relationship between chlorophyll *a* and TN or TP.

2.3.2.3 Balanced Faunal Communities

The health of estuarine and coastal biological communities, from fish to benthic macroinvertebrates to plankton, depends critically on sufficient DO (e.g., Diaz 2001; Diaz and Rosenberg 2008). In estuaries and coastal waters, low DO is one of the most widely reported consequences of nitrogen/phosphorus pollution and one of the best predictors of a range of biotic impairments (e.g., Bricker et al. 2003a; Bricker et al. 1999). Low DO causes impacts to marine life ranging from mass cross-species mortality to chronic impairment of growth and reproduction. Thus, DO is a measurement endpoint proxy for a wide range of marine life for which DO requirements for survival, growth, and reproduction are known.

Estuaries may exhibit large, diurnal DO concentration swings characterized by high concentrations during the daylight hours, followed by periods of low concentrations (potentially hypoxic or anoxic) during the nighttime period. Furthermore, highly productive systems tend to have large amounts of detritus that settle to the bottom sediments and are oxidized by bacteria, further consuming oxygen and resulting in sediment nutrient releases (Cloern 2001).Water column stratification due to salinity and/or temperature gradients reduces the mixing of oxygen-rich surface waters (where oxygen is transferred into the water from the atmosphere) with oxygen-poor bottom waters (where oxygen is consumed through sediment diagenesis processes), exacerbating the effects of low DO on bottom-dwelling species.

In the case of DO, the State of Florida has an established a DO standard for estuarine and coastal waters that states the average DO shall not be less than 5.0 mg/L in a 24-hour period and shall never be less than 4.0 mg/L, with normal daily and seasonal fluctuations maintained (F.A.C. 62-302.530). Subsequently, numeric criteria for TN and TP can be computed using the relationship between DO and TN or TP.

2.3.3 Water Quality Indicator Variables for Expressing Criteria

Based on EPA guidance (USEPA 2001) and an assessment of the available literature, the numeric criteria for Florida's estuaries, coastal waters and southern inland flowing waters will address the following three indicator variables: TN concentration (as mg/L), TP concentration (as mg/L), and chlorophyll *a* concentration corrected for pheophytin (chlorophyll *a* as µg/L). Appropriate numeric criteria for these three variables will help ensure protection of the biological assessment endpoints identified in Figure 2-1 are achieved, thereby supporting a balanced natural population of aquatic flora and fauna.

While the conceptual model of eutrophication continues to evolve (Cloern 2001), it is clear that nitrogen and phosphorus are the primary macronutrients that enrich waters and can cause nuisance levels of algae (Elser et al. 2007; Howarth et al. 2002). Conditions that allow phytoplankton to accumulate (i.e., adequate light, optimum velocity or mixing, low loss to grazing, etc.) will not result in high biomass without sufficient nutrient supply (USEPA 2001). Often the addition of both nitrogen and phosphorus will elicit greater phytoplankton biomass stimulation than the sum of both nutrients added separately (Fisher et al. 1992). There are reported cases where both nitrogen and phosphorus are required to elicit a phytoplankton biomass production response in estuaries (Flemer et al. 1998), suggesting that nitrogen and phosphorus supply rates were equally limiting. On the other hand, tropical lagoons, with carbonate sands low in phosphorus and unaffected by human activity, are prone to phosphorus limitation. For example, the seagrass *Thalassia testudinum* was found to be phosphorus-limited in Florida Bay (Powell et al. 1989; Fourqurean et al. 1992a; Fourqurean et al. 1992b).

2.3.3.1 Total Nitrogen

Nitrogen is an important limiting nutrient of algal biomass production (USEPA 2001). TN consists of organic and inorganic forms. Stimulated algal biomass production is typically attributed to inorganic nitrogen (Stepanauskas et al. 1999), although some dissolved organic nitrogen may be used for algal growth (dissolved and particulate organic nitrogen are involved in recycling processes) (USEPA 2001). In estuaries, nitrogen concentrations, especially the inorganic forms, typically vary widely seasonally, interannually, and along salinity gradients (USEPA 2001). In those estuaries where nitrogen has been demonstrated to limit algal biomass production, it typically does so at higher salinities along the salinity gradient. Denitrification may remove from a few to approximately 50 percent of the TN load entering temperate estuaries annually (Seitsinger 1988; Cornwell et al. 1999) depending largely on residence time of the water, sediment biogeochemical conditions (e.g., benthic macrofauna present to maintain irrigation, oxic conditions in the overlying bottom water), and water column depth. This process helps to modulate extreme dissolved inorganic N concentrations (USEPA 2001).

2.3.3.2 Total Phosphorus

Phosphorus is often the nutrient that most limits algal production in tidal fresh to oligohaline areas of estuaries as well as areas with a wider range of salinity in certain subtropical to tropical marine systems (USEPA 2001). Phosphorus occurs in natural waters and in wastewaters almost solely as phosphates. These are classified as orthophosphates, condensed phosphates, and organically bound phosphates. Common analytes are TP and dissolved or particulate organic phosphorus (DOP, POP). These compounds may be soluble, in particulates or detritus, or incorporated as organic phosphorus in organisms. Phosphorus is essential to the growth of organisms and can limit phytoplankton biomass production, which is most commonly observed in freshwater systems (Hecky and Kilham 1988) and some estuaries and coastal marine systems. In instances where phosphate is limiting, the discharge of raw or untreated wastewater, agricultural drainage, or certain industrial wastes may stimulate the growth of algae (USEPA 2001). Some fraction of phosphorus may be strongly embedded in a mineral matrix, rendering that fraction relatively inert to biological utilization except by algae that have the capability to break down DOP with alkaline phosphatase (algal and free phosphatases) and utilize the phosphate as inorganic phosphate (Huang and Hong 1999).

2.3.3.3 Chlorophyll *a*

Chlorophyll *a* is a measure of phytoplankton abundance (or biomass) in the water column and can serve as an index of the productivity and trophic condition of waters. Chlorophyll *a* biomass reflects the *standing stock*, which is the balance of growth and loss in pelagic waters. The benefits of chlorophyll *a* as an indicator variable are its relevancy to conditions of Florida's ecosystems, its sensitivity to stressors such as nutrients, and ease of monitoring (Boyer et al. 2009). Elevated concentrations of chlorophyll *a* are indicative of enhanced phytoplankton production. Excess primary production can cause a variety of negative effects (Bricker et al. 2003b; Vitousek et al. 1997). For example, excess primary production can reduce water clarity, resulting in reduced light availability for benthic algae, macrophytes, and seagrasses (Boyer et al. 2009; Bricker et al. 2008). Seagrass decomposition and destabilization of sediments may then result in more nutrient inputs into the water column from sediments (Boyer et al. 2009). Excess production of chlorophyll *a* also provides greater loads of reduced carbon, which fuels respiration, decreases DO, and results in hypoxic and anoxic conditions (Vitousek et al. 1997).

2.3.3.4 Not Selected for Numeric Criteria Development

Based on the reviewed literature, EPA is not considering using the following nutrient-sensitive biological assessment endpoints to translate Florida's narrative criterion into numeric values: (1) HABs, (2) coral, (3) epiphytes, (4) macroinvertebrate and fish indices, (5) macroalgae, (6) Spartina marshes (salt-marshes), and (7) Eastern oysters (*Crassostrea virginica*). In general, these assessment endpoints have not been selected because there is either an absence of sufficient data to assess the effects of measured nitrogen/phosphorus concentrations, or there is an alternative sensitive assessment endpoint available that is a better indicator of nitrogen/phosphorus pollution. Appendix B provides additional information regarding the scientific rationale for not including these organisms in development of numeric criteria for estuaries and coastal areas.

2.4 Potential Data Sources

EPA has assembled a large and diverse resource of environmental data to support the analytical approaches that the Agency is considering for development of numeric criteria for Florida waters. This has been accomplished with the active assistance of state and local governmental agencies in Florida, including the Florida Department of Environmental Protection, the Florida Fish and Wildlife Research Institute, Florida's Water Management Districts, and several county governments. Significant data have also been provided by multiple Federal agencies, including the USGS, NASA, and NOAA, as well as public and private research institutions. Data have been provided to EPA both via existing online data portals and other means (e.g., e-mail, FTP, mail). The magnitude of data available reflects in part Florida's substantial investment in data collection and data management, which EPA noted in its 2009 determination that new or revised water quality standards were needed (see Section 1.3). The substantial quantity of data also reflects ongoing data collection by Federal agencies.

The many different types of data that EPA is considering reflect both the richness of data that are available and the many types of data that could be needed to support the analytical effort that EPA is considering. Because EPA is considering water quality simulation models as one of the analytical approaches, the number of different kinds of data that could be needed is very broad and extends well beyond water quality monitoring data.

The paragraphs below describe in further detail major data sets that EPA may use, the sources of the data including internet sources for the data or information about the data, and which aspects of criteria development the data may support.

Waterbody Delineation. EPA is considering as a source of information delineating waterbodies in Florida, the Waterbody Identification number (WBID) GIS layer. The WBID layer is available via the FDEP GIS portal (http://www.dep.state.fl.us/gis/datadir.htm).

Water Quality Monitoring Data. EPA is considering water quality monitoring data from a variety of sources. These data may be used in almost every aspect of criteria development and pertain to both freshwater and marine water quality. The largest source of water quality data is the Florida Department of Environmental Protection's Impaired Water Rule Database, version 40 (IWR40). FDEP maintains the IWR database and updates it quarterly with data it identifies and evaluates from trusted sources throughout Florida. The database is available from FDEP at http://publicfiles.dep.state.fl.us/dear/IWR/. Additionally for Chapter 5, the Southeast Environmental Research Center (SERC) Water Quality Monitoring Network is a significant source for more than a decade of quarterly or monthly (depending on the location) water quality data for South Florida, including the Biscayne Bay, Florida Bay and the Florida Keys (http://serc.fiu.edu/wqmnetwork). Further water quality and other data specific to South Florida is available from DBHYDRO, maintained by the South Florida Water Management District (http://www.sfwmd.gov/portal/page/portal/xweb%20environmental%20monitoring/dbhydro%20application). Water quality data for South Florida canals has been provided to EPA by the Miami-Dade Department of Environmental Resources Management and Broward County Department of Planning and Environmental Protection. Many other counties in Florida conduct water quality monitoring and a significant quantity of these data are already integrated by FDEP into the IWR database.

Land Use Data. EPA is considering land use data as inputs to water quality simulation models (i.e., mechanistic watershed models), as described in Chapters 3 and 6. EPA is also considering land use data as part of a Landscape Development Intensity Index calculation in Chapter 5. Since several Florida watersheds extend into Alabama and Georgia, EPA is considering sources of land use data from all three states. EPA is considering land use data for Florida based on 2005 imagery as reported by FDEP and various Water Management Districts. Data for Georgia are from Georgia Land Use Trends (GLUT) 2005 (http://narsal.uga.edu/glut.html). EPA is considering the 2005 National Land Cover Database for data from Alabama (http://www.epa.gov/mrlc).

Meteorological Data. EPA is considering meteorological data, including rainfall and wind speed and direction as inputs to mechanistic watershed models and hydrodynamic and water quality models for estuaries, as described in Chapters 3 and 6. EPA is considering data from approximately 120 sites in Florida. These data may be obtained from the National Climatic Data Center, which reports data for more than 1800 stations in Florida, Georgia, and Alabama. Further information is available at http://www.ncdc.noaa.gov.

General Hydrology. EPA is considering hydrology data, including the National Hydrography Dataset Plus (NHDPlus, http://www.horizon-systems.com/nhdplus), which provides subwatershed and flow line delineations that can be used in watershed models (Chapter 3 and 6). EPA is considering stream discharge data and flow velocity data from the US Geological Survey, available through the National Water Information System (NWIS, http://waterdata.usgs.gov/nwis). These data could be used to parameterize, calibrate and evaluate mechanistic watershed models (Chapter 3 and 6) and identify canals (Chapter 5). EPA is considering use of the National Elevation Dataset 1/3 arc-second (10 meter by 10 meter) for computing elevations and slopes. The National Elevation Dataset is available from the USGS (http://ned.usgs.gov/). EPA is also considering water surface elevation data from NOAA tide gauges for use as boundary conditions, calibration, and evaluation data for hydrodynamic models (Chapter 3). These data are reported by NOAA's Center for Operational Oceanographic Products and Services (http://www.tideandcurrents.gov). EPA obtained bathymetric data for Florida estuaries and coastal areas from the NOAA National Geophysical Data Center (http://map.ngdc.noaa.gov).

NPDES Point-Sources and Water Withdrawals. For use in water quality simulation models (Chapter 3 and 6) EPA is considering data on NPDES-permitted point sources and water withdrawals obtained from Florida's Waterwater Facility Regulation (WAFR) system (http://www.dep.state.fl.us/water/wastewater/facinfo.htm).

Ocean Color Satellite Data and Field Validation. EPA is considering data from two NASA satellite-borne ocean color sensors for use in development of numeric criteria for offshore coastal waters in Florida (Chapter 4). These sensors include the Sea-viewing Wide Field-of-view Sensor (SeaWiFS) and the Moderate Resolution Imaging Spectroradiometer (MODIS). The period of record for both sensors is more than 10 years. EPA identified at least six sources of shipboard data to compare with satellite data. EPA is considering data from the Northeastern Gulf of Mexico (NEGOM) project (NOAA National Oceanographic Data Center, http://www.nodc.noaa.gov), the Ecology and Oceanography of Harmful Algal Blooms project (ECOHAB), as well as collections of data from the Mote Marine Laboratory (http://www.mote.org), Florida Fish and Wildlife Research Institute (http://research.myfwc.com/), and SeaWiFS Bio-optical Archive and Storage System (SeaBASS, http://seabass.gsfc.nasa.gov/).

Seagrass Coverage Layers. EPA is considering extensive datasets describing historical coverage of submerged aquatic vegetation in Florida estuaries (Chapter 3). Data were provided to EPA by FDEP, the US Geological Survey Wetland Research Center (http://sdms.cr.usgs.gov/pub/flsav.html), Florida's water management districts, and by the Tampa Bay National Estuary Program (see Table 3-6).

Other Data. EPA may consider other well-documented data obtained from known sources. For example, EPA is considering water quality data, water level data, meteorological data, and current velocity data collected for Pensacola Bay in 1996-2000, 2002-2004 and 2009 by EPA's National Health and Environmental Effects Research Laboratory, Gulf Ecology Division. EPA has provided the water quality data to FDEP for inclusion in the IWR database.

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3 Numeric Nutrient Criteria Development in Florida Estuaries

An estuary is a part of a stream[5] or other body of water that has an unimpaired connection with the open sea and where sea water is measurably diluted with freshwater derived from land drainage. In order to translate Florida's current narrative nutrient criterion, EPA is considering developing numeric criteria for nitrogen/phosphorus pollution in Florida's estuaries on a system-specific basis. A system-specific approach would allow the Agency to consider the individual characteristics of these estuarine ecosystems in groups with common characteristics. For example, water quality and biological communities in estuaries are affected by a combination of basin shape, tides, and the magnitude, location, and quality of freshwater inflows. The semi-enclosed basins that define the spatial extent of estuaries areas may also create sub-regions within estuaries with differentiated water quality and assessment endpoints. Thus, EPA is considering approaches that may result in numeric criteria specific to sub-regions within estuaries.

This chapter describes the approaches EPA is considering to derive numeric criteria for estuarine waters in Florida (Figure 3-1), exclusive of marine waters in South Florida, which are addressed in Chapter 5. We describe the approach that EPA is considering for delineating estuaries into discrete areas for the purpose of organizing the criteria development process. We also discuss the concepts of assessment endpoints and indicator variables, and the specific endpoints and indicators that EPA is considering for use in development of numeric estuarine criteria. We discuss the rationale that may be used for selecting specific water quality indicator variables for which EPA may develop criteria. Finally, we discuss three approaches: (1) reference conditions, (2) stressor response relationships (regression models), and (3) water quality simulation modeling that EPA could use independently or in combination to develop numeric estuarine criteria.

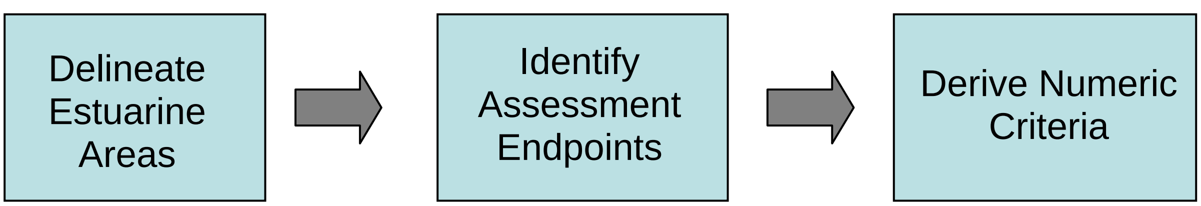

Figure 3-1. Flow chart for the development of numeric criteria for Florida estuaries. The approach EPA is considering will involve delineating the state's estuarine waters into discrete estuarine areas. Following an overall framework approach, aquatic life endpoints and specific data and methods will be selected on a system-specific basis. Numeric criteria for each estuary, or sub-segment of each will be derived using those data and methods.

[5] For the purpose of this effort, a stream has been defined as free-flowing, predominantly fresh surface water in a defined channel, and includes rivers, creeks, branches, canals, freshwater sloughs, and other similar water bodies. Predominantly fresh waters have been previously defined as surface waters in which the chloride concentration at the surface is less than 1500 mg/L (salinity less than ~2.7 psu). EPA is considering alternative definitions which could be based on conductivity or salinity.

3.1 Delineating Estuaries

The first step in any approach for developing numeric criteria for nitrogen/phosphorus pollution is delineating the water bodies. Delineating the estuarine waters provides an organizational framework for developing and presenting the scientific approach, applying the methods and approaches most appropriate to each estuary, and ultimately deriving criteria. For estuaries outside of South Florida, EPA is considering a delineation approach based on the natural geographic limits of estuarine basins and their associated watersheds. Natural constrictions between estuarine basins tend to limit water flow and exchange between estuaries, even if exchanges are not eliminated entirely. This approach results in 23 estuarine areas (Figure 3-2), including five in the Florida panhandle region, six in the Big Bend region, four in southwest Florida, and eight on the Atlantic coast.[6] This general approach has been utilized previously in development of the NOAA Coastal Assessment Framework (Bricker et al. 1999). EPA is presenting the approaches that it is considering for delineating marine waters in South Florida separately (see Chapter 5) because the flows in the waters in that part of the State are heavily managed and natural boundaries between basins are less defined.

EPA is also considering utilizing natural geographic boundaries as an approach for delineating sub-segments within estuaries. FDEP has established a waterbody identification scheme (WBIDs) which defines segments within estuaries (see Figure 3-3). This approach is based to a significant extent on natural geographic boundaries within and among Florida's estuaries, but it has been modified by the State over time for a variety of reasons and could be subject to change in the future. Thus, EPA is considering modifying the boundaries of FDEP's WBIDs to achieve the objective of homogenous water quality within segments while maintaining a reasonable spatial scale for criteria development (e.g., not an excessive number of very small segments).

[6] A total of 30 estuarine and coastal areas are identified. EPA's approaches for offshore coastal waters and for South Florida waters are presented in separate chapters.

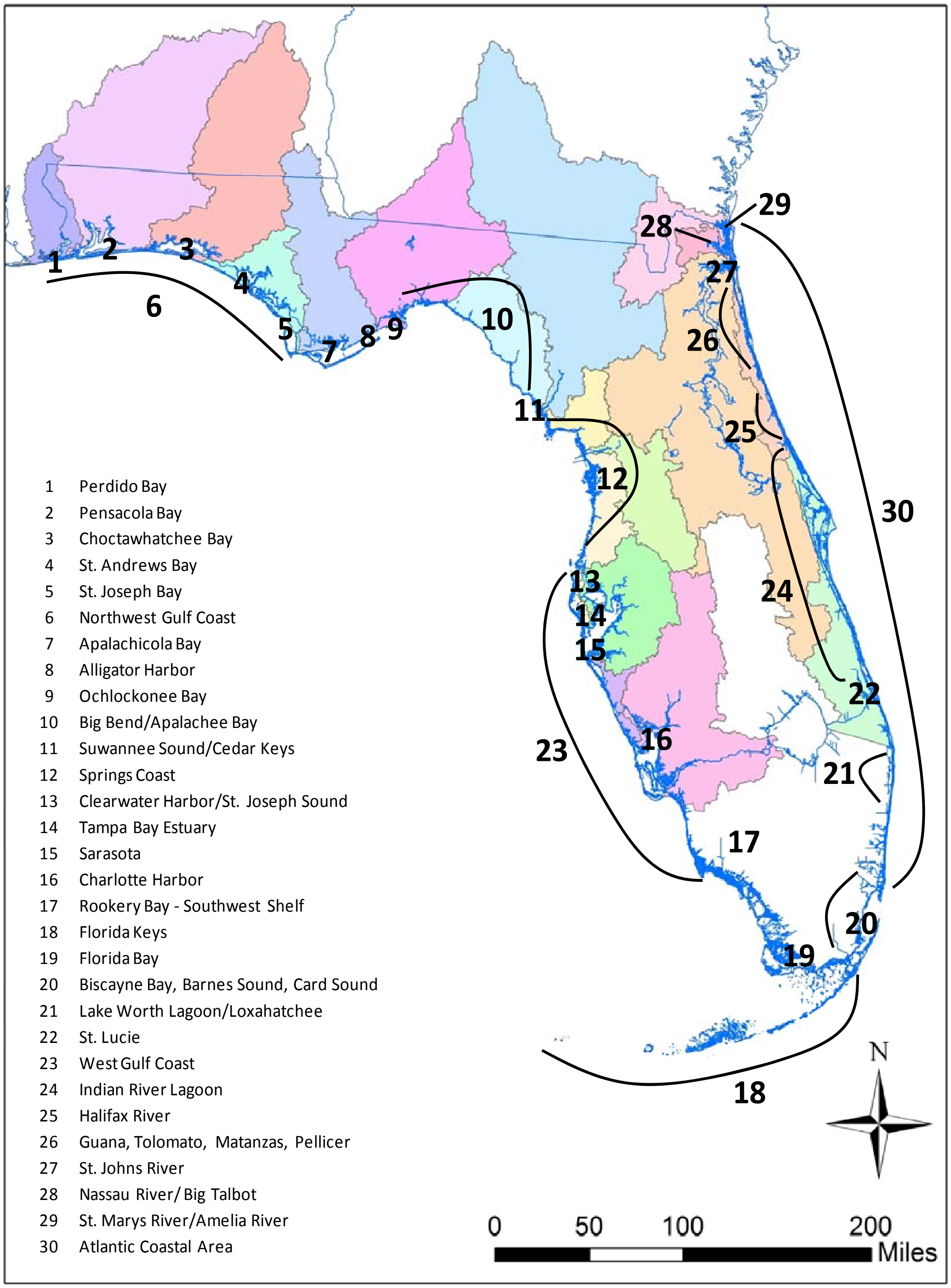

Figure 3-2. Major areas of estuarine and coastal waters in Florida that EPA has identified for the purpose of organizing development of numeric criteria. The approach for developing numeric criteria for offshore coastal waters of the northwest Gulf coast [6], west Gulf Coast [23], and Atlantic Coast [30] is described in Chapter 4. Approaches for South Florida [17, 18, 19, 20] are described in Chapter 5. The approach for the remaining 23 areas is described in this chapter. Shading of land areas illustrates watershed areas to be used in EPA's proposed framework for estimating watershed loadings of TN and TP to estuaries using a water quality simulation model (Section 3.3.3)

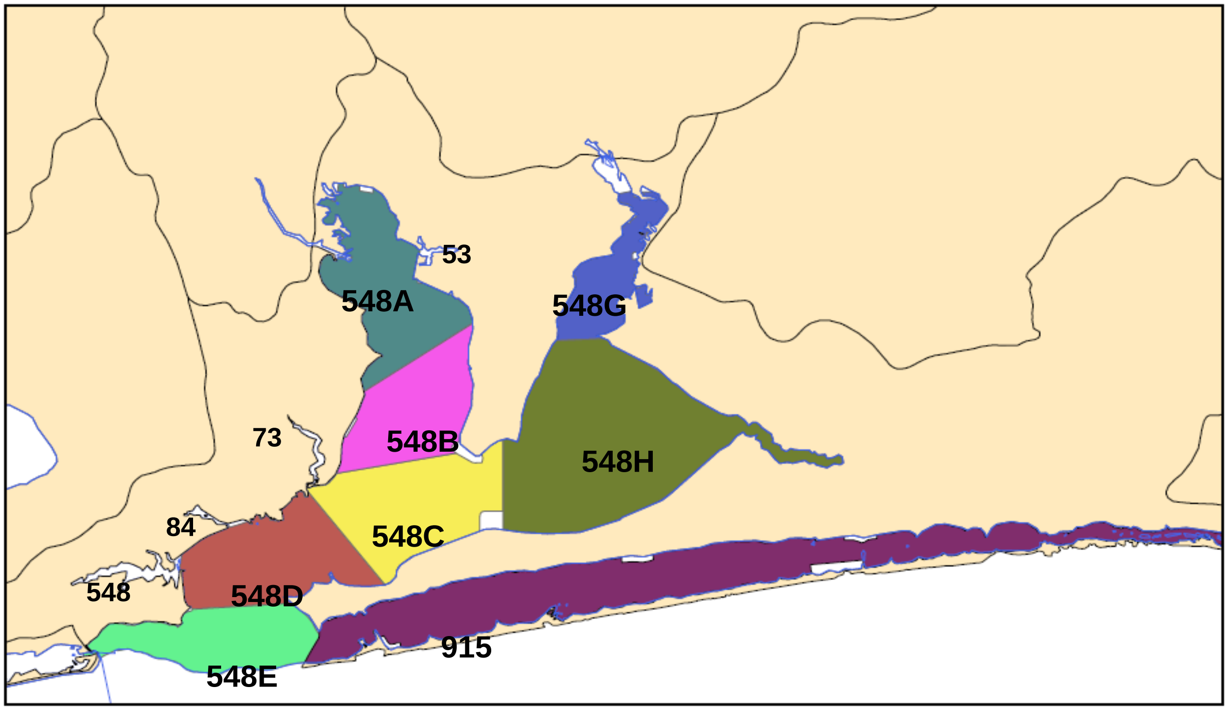

Figure 3-3. Map of Pensacola Bay illustrating the boundaries associated with FDEP's WBIDs for Pensacola Bay. Shaded segments account for the majority of the estuarine area. Additional segments are identified for smaller creeks and bayous that adjoin the main part of the Bay. Boundaries for major Bay segment are generally associated with geographic features of the basin, with segments occurring at regular intervals to ensure relatively homogeneous water quality within sub-segments of the Bay.

3.2 Selection of Assessment Endpoints and Indicator Variables

Biological assessment endpoints for numeric criteria development are specific, ecologically important, and nutrient-sensitive statements of the environmental value to be protected, in this case to translate Florida's narrative nutrient criterion and support a balanced natural population of aquatic flora and fauna within Florida estuaries. In environmental risk assessment (ERA) guidance these are referred to as "assessment endpoints." Indicator variables are quantitative measures that point to or "indicate" the status of the assessment endpoint in a waterbody, or in ERA terms "measures of effect" (or "measurement endpoints"), "measures of exposure," and "measures of ecosystem and receptor characteristics." The TMDL and Watershed Approach literature tends to refer to these measures as "indicators." To be a useful indicator variable, the indicator must bear a defensible conceptual relationship to the endpoint. Data must be available to measure or otherwise quantify the indicator value in water bodies to which it will be applied. It should also be possible to relate the value of indicator variables in some way to nitrogen/phosphorus concentrations and/or nutrient loadings. Selection of assessment endpoints is a critical early step that defines many subsequent aspects of criteria development. EPA has reviewed the scientific literature to identify candidate endpoints that are ecologically important, widely applicable in Florida estuaries, and sensitive to nutrients. As mentioned in Chapter 2, three assessment endpoints that EPA is considering for use in the derivation of numeric criteria in estuaries are discussed below. EPA is considering one or more indicator variables for each of these endpoints. Please refer to section 2.3 for more detail regarding these assessment endpoints and indicator variables.

Healthy seagrass communities. This assessment endpoint is related to habitat and encompasses seagrass condition (e.g., density), species composition, and distribution approximating local historical conditions.[7] EPA is considering the depth of colonization of seagrasses as a readily quantifiable indicator of seagrass condition. For a number of estuaries, historical values of the depth of colonization can be used to establish values that support the designated use. Moreover, depth of colonization and areal extent of seagrass are related via the bathymetric profile (e.g., Janicki and Wade 1996). Depth of colonization of seagrasses is a useful assessment endpoint for numeric criteria development because water clarity, which affects the depth of colonization is affected[8] by phytoplankton biomass (measured as chlorophyll *a*) and nutrient (TN and TP) loading.

Balanced phytoplankton biomass and production. This assessment endpoint is characterized by normal levels of biomass and productivity of phytoplankton communities, with an absence of harmful or nuisance species or other adverse shifts in species dominance, and no unusual proliferations of macroalgae or epiphytic algae. EPA is considering using chlorophyll *a* as an indicator of the changes in phytoplankton species composition. EPA is considering this endpoint because proliferations of macroalgae, epiphytes, or nuisance algal species may be the most apparent nutrient pollution (TN and TP) impacts in some estuaries, particularly those where seagrasses are not normally present or where low dissolved oxygen is unlikely due to physical factors (e.g., strong tides).

Balanced faunal communities. This assessment endpoint is refers to communities of benthos (bottom-dwellers), plankton (drifting organisms), and nekton (swimming organisms, such as fish) exhibiting healthy levels of biomass and production. Low dissolved oxygen resulting from nutrient pollution is a key mechanism by which nutrients may impact this assessment endpoint. Because quantitative indicator variables for this endpoint are lacking, EPA is considering using the absence of hypoxia (i.e., absence of low dissolved oxygen), as an indicator of balanced communities. Dissolved oxygen is a useful indicator for numeric criteria development because ecological modeling approaches can be used to predict the response of dissolved oxygen to nutrient pollution loading (TN and TP).

3.3 Numeric Criteria Approaches

EPA is considering three basic categories of approaches to derive numeric criteria for Florida estuaries. These approaches include (1) reference condition approaches, (2) stressor response relationships, and (3) water quality simulation models. Associated with each category of approach are specific strengths and weakness, factors indicating that it could be used, and factors that indicate another approach may be needed (Table 3-1). EPA will consider these as it

[7] Seagrasses are naturally absent from some northeast Florida estuaries. For those estuaries, water clarity requirements to support maintenance of seagrasses are not being considered as an endpoint.
[8] Water clarity in estuaries is largely determined by the sum of light attenuation caused by phytoplankton biomass, of which chlorophyll a concentration is an indicator, non-pigmented suspended solids, and colored dissolved organic matter (e.g., Gallegos 1994, 2001).

determines which approaches should be used given the ecological details pertinent to each estuary as well as the different types and quantities of data available. EPA could consider several different types of models and information to derive numeric criteria for different estuaries and could consider simultaneously more than one type of information for a single estuary.

Table 3-1. Strengths, weaknesses, indications (situations where approach is most applicable), and contraindications (situations where another approach may be needed) for each of the three categories of criteria development approaches that EPA is considering

	Strengths & Weaknesses	Most Applicable When	Least Applicable When
Reference Condition Approaches	*Strengths* • Simple, direct and understandable; provides information to quantify criteria. *Weaknesses* • Need quantitative data to characterize the reference condition that reflects support of the designated use.	• Substantial water quality data are available and the estuary is minimally impacted by nitrogen/phosphorus pollution sources. • Substantial water quality data are available from a historical period when the estuary was minimally impacted by nutrients. • The estuary is very similar to another estuary to which one of the above conditions applies.	• The estuary is impacted by nitrogen/phosphorus pollution sources and is likely impaired by nutrients. • Little or no data are available from a historical period when the estuary was not minimally impacted by nutrients • The estuary is considered relatively unique.
Stressor Response Relationships (Regression Models)	*Strengths* • Easy to understand and visualize; uncertainty may be quantified, provides linkage between criteria and aquatic life uses, can quantify relationships between different criteria values. *Weaknesses* • Regressions can be affected by covariates; may not address additive or interacting effects of more than one causal factor.	• Extensive data are available, spanning multiple years and spanning a range of nutrient loading rates and water quality response. • Simple regression relationships exist and quantify relationships between nutrient loading and/or nutrient concentrations and water quality responses. • Response is consistent across many estuaries	• Little or no data are available • Complex relationships between nutrients and water quality responses involve multiple interacting causes, including physical-biological coupling. • Key ecological processes and interactions are different or unique compared to other estuaries.
Water Quality Simulation Models	*Strengths* • Can provide detailed simulation results for many variables, addressing magnitude, frequency and duration; addresses physical-biological coupling. *Weaknesses* • May not address important ecological processes; many unknown model parameters including boundary conditions; may not be valid for unobserved conditions.	• Important ecosystem processes are well-understood • Available data are from process studies or other isolated studies, rather than consistent monitoring over multiple years. • Interactions are complex, involve physical-biological interactions, or are spatially structured. • Relatively little site-specific data are available.	• Mechanisms governing interaction among nutrient sources, water quality, and biological responses are not well understood. • Critical inputs to model are completely unknown (e.g., large open boundaries) • Linkages between possible model outputs and use attainment are not well-defined. • Adequate data are not available as model input.

3.3.1 Reference Condition Approaches

Reference condition approaches can take a variety of forms, defined by the source of the reference condition. EPA has previously recommended (e.g., USEPA 2000c) that a percentile of water quality measurements in a sample of minimally-impacted waterbodies, which are known to be fully supporting designated uses (i.e., not impaired), could serve as numeric criteria in similar waterbodies. In this case, a reference condition is derived from a reference population of waterbodies. In Florida, there are a small number of estuaries (i.e., 23 estuarine systems compared to >1,000 streams and lakes in Florida) that leads EPA to consider developing system-specific criteria for estuaries. EPA is considering a reference condition approach to be most applicable when (1) historical data adequately describe water quality conditions when the estuary was minimally-impacted by nitrogen/phosphorus pollution and was supporting balanced natural populations of aquatic flora and fauna (i.e., historical reference condition) or (2) when the estuary is currently minimally-impacted by nitrogen/phosphorus pollution and currently supporting balanced natural populations of aquatic flora and fauna (i.e., current-conditions reference condition). In either case, interpretation of the status of reference conditions could be based on examining the assessment endpoints and associated indicators that EPA has identified (Section 3.2). EPA has not yet identified any impaired estuaries for which historical water quality data could provide a suitable reference condition. Therefore, EPA is largely considering using a reference condition approach only in the second case, namely when current water quality conditions are supporting balanced natural populations of aquatic flora and fauna.

To evaluate assessment endpoints and associated water quality indicator variables, EPA will consider data from FDEP's Impaired Waters Rule (IWR) database. The number of TN, TP, and chlorophyll *a* observations from the IWR are presented in Table 3-2. EPA will also consider data from peer-reviewed literature, reports, and other data sources. A complete listing of the data sources EPA is considering can be found in Section 2.4, Potential Data Sources. To derive criteria from current water quality conditions, EPA is considering computing two statistical reference points from the water quality observations. These could include (1) an average or median concentration and (2) an upper percentile concentration. By simultaneously considering both an indicator of central tendency and a measure of higher concentrations, the criteria could ensure that future water quality conditions remain similar to present conditions (i.e., the conditions associated with support of balanced natural populations of aquatic flora and fauna). As an alternative, EPA is also considering whether it should utilize annual geometric mean water quality. Water quality observations in the future could be compared with current water quality using this binomial approach (see Section 5.6.1.2).

Table 3-2. Estuary water quality data inventory for causal (TN and TP) and response variables (chlorophyll *a*) from FDEP Impaired Waters Rule database. Count is the number of observations; years represent the time period over which the observations were collected.

Estuary		TN		TP		Chlorophyll *a*	
		Count	Years[a]	Count	Years[a]	Count	Years[b]
1	Perdido Bay	241	1999–2010	268	1999–2010	2,591	1974–2010
2	Pensacola Bay	1,530	1998–2010	1,597	1998–2010	9,278	1968–2010
3	Choctawhatchee Bay	1,827	1997–2009	1,972	1997–2009	4,828	1974–2009
4	St Andrew Bay	1,234	2000–2009	1,295	2000–2009	3,910	1973–2010

5	St Joseph Bay	20	2008–2009	20	2008–2009	100	1974–2009
7	Apalachicola Bay	476	2000–2007	278	2000–2007	728	1974–2007
8	Alligator Harbor	N/A		N/A		4	1987–1987
9	Ocklockonee Bay	25	2000–2007	27	2000–2007	57	1975–2007
10	Big Bend/Apalachee Bay	158	2000–2009	173	2000–2009	228	1973–2009
11	Suwannee Sound/Cedar Keys	155	2000–2009	166	2000–2009	731	1995–2009
12	Springs Coast	320	2002–2009	322	2002–2009	868	1982–2009
13	Clearwater Harbor/ St. Josephs Sound	722	2002–2009	695	2002–2009	1,810	1991–2009
14	Tampa Bay	12,844	1996–2009	13,532	1996–2009	58,891	1971–2009
15	Sarasota Bay	5,463	1997–2009	5,460	1997–2009	7,911	1978–2009
16	Charlotte Harbor	19,319	1996–2009	19,177	1996–2009	19,990	1973–2009
21	L. Worth Lagoon/ Loxahatchee Estuary	674	1997–2009	777	1997–2009	1,499	1974–2009
22	St. Lucie Estuary	7,830	1996–2008	7,708	1996–2008	17,097	1973–2009
24	Indian River Lagoon	8,191	1999–2010	10,912	1999–2010	37,429	1973–2010
25	Halifax River	986	1999–2009	996	1999–2009	6,318	1973–2009
26	Guano/Tolomato/Matanzas/ Pellicer	1,880	1999–2010	1,897	1999–2010	3,941	1973–2010
27	St. Johns River Estuary	4,240	1996–2008	4,241	1996–2008	7,575	1973–2009
28	Nassau River/ Big Talbot	391	1998–2009	244	1998–2009	193	1973–2008
29	St. Marys River/Amelia River Estuary	78	1998–2004	79	1998–2004	127	1973–2004

a. Data from IWR Run 40 only; data prior to 1996 have not yet been combined with IWR Run 40 data. b. Data from IWR Run 40 combined with data from prior IWR runs. N/A = no data available.

3.3.2 Stressor Response Relationships

Regression models usually express a stressor response relationship between one or more explanatory variables and a single response variable. Regression models can also encompass more complex linear statistical models such as analysis of covariance models (i.e., models involving both continuous and categorical explanatory variables), as well as non-linear regression models.

Two major strengths of regression models as approaches that could be used for development of numeric criteria are that they are closely grounded in environmental data and, in the case of a single explanatory variable, easy to communicate, often by simple graphics (e.g., bi-variate plots). Accordingly, they can be easy to understand and less dependent upon assumptions and other analytical decisions made by investigators (Table 3-1). Additionally, statistical methods for fitting regression models often permit estimation of limits of uncertainty for predictions, even for complex regression models (e.g., Hoos and McMahon 2009). Regression models require adequate data to develop. Additionally, other environmental variables that covary with explanatory variables of interest can introduce uncertainty in estimates of regression model parameters. It may also be difficult to find highly predictive regression models for complex ecological systems that include many interacting factors that impact the dependent variable. This is especially true when important processes occur on differing temporal and spatial scales. However, useful regression models do exist and have been applied successfully to quantify relationships among water quality indicator variables in estuaries.

Examples of regression models that could be useful for development of numeric criteria include (1) models relating a "causal variable" such as TN or TP loading or concentration to a response variable such as chlorophyll *a*, (2) models relating TN or TP loading to average concentration in estuarine waters, and (3) models quantifying relationships between other environmental variables. Several regression models could be utilized in combination to derive numeric criteria. EPA is evaluating existing regression models and is considering work to identify more regressions that would be useful for development of numeric criteria in Florida.

Janicki and Wade (1996) describe a well-known application of regression models for development of nutrient loading limits for Tampa Bay, which EPA is considering as a useful example of the application of regression models for development of numeric criteria. This approach targets the areal extent of seagrass as an indicator of healthy seagrasses in Tampa Bay and of overall attainment of balanced natural populations of aquatic flora and fauna. Targets for areal extent were based on historical seagrass distributions. Empirical relationships were used to quantify the relationship between areal extent of seagrass and depth of colonization of seagrass in Tampa Bay. Depth of colonization estimates for each segment of Tampa Bay and empirically determined estimates of the light requirements of seagrasses were used to compute segment-specific limits for average light attenuation coefficient needed to support seagrasses at target depths. Regressions were developed to quantify the relationship between light attenuation coefficient and average chlorophyll *a*, and the relationship between average chlorophyll *a* and TN loading to each segment of the Bay. In combination, these relationships provide an approach for developing numeric criteria for TN loading (causal variable) and average chlorophyll *a* (response variable) for each segment of Tampa Bay. As described by Janicki and Wade (1996), the approach does not address assessment endpoints other than seagrasses, TP loading, or ambient concentrations of either TN or TP in estuarine waters. However, EPA is considering this approach as a potentially useful example of the application of regression models for development of numeric criteria for Florida estuaries that could be applied using other types of models, such as water quality simulation models (Section 3.3.3), to quantify relationships between water quality variables.

Whereas Janicki and Wade (1996) describe an approach for numeric criteria development on a highly system-specific basis, Steward and Lowe (2010) illustrate another regression-based approach that emphasizes the similarity among estuaries in their response to nutrients. Specifically, Steward and Lowe (2010) found that estimates of protective TN and TP loading rates for a variety of estuaries and other waterbody types, derived independently using a variety of approaches could be related via the average residence time for freshwater (Figure 3-4). This relationship could be used to compute an estimate of the protective TN and TP loading rate for any estuary, given only an estimate of residence time. Although EPA is considering this approach as a tool that could be applied in combination with other information to develop and evaluate nutrient loading targets, EPA may not apply this model as the principal evidence for developing a nutrient loading target for an estuary because it does not provide estimates of protective loads with sufficient precision. This may not be fully evident in Figure 3-4 because of the log scale. Although the model has a high coefficient of determination and narrow limits of uncertainty, accurate predictions of loading limits depend on estimates of average residence time, which can be difficult to estimate precisely. In addition, EPA must carefully consider the appropriate interpretation of the relationships described by Steward and Lowe, which are presented as estimates of the loading levels associated with mesotrophy. For some estuaries in Florida, a mesotrophic condition may not support balanced natural populations of aquatic flora and fauna, or maintain the current chemical and biological integrity. Finally, although the outstanding coefficient of determination for the relationships could lead one to conclude that they reflect important new insights, the strength of the relationships is a consequence of scaling. Given the very narrow range in appropriate values for average TN or TP concentrations (~ 3 to 4-fold variation) compared to a very large range in residence times (~1 day to 3 years) and protective loading estimates (1 to 300 g N/m^2/y), a strong correlation is likely, regardless of the average concentration determined to be associated with a mesotrophic condition. The fact that the estimated slopes are generally not equal to -1 (i.e., -0.89) could reflect either the fact that the residence time for nutrients is expected to be less than for water (i.e., due to denitrification and other nutrient losses; Dettmann 2001) or alternatively, could reflect a slight increase in average depth among estuaries with longer residence times.

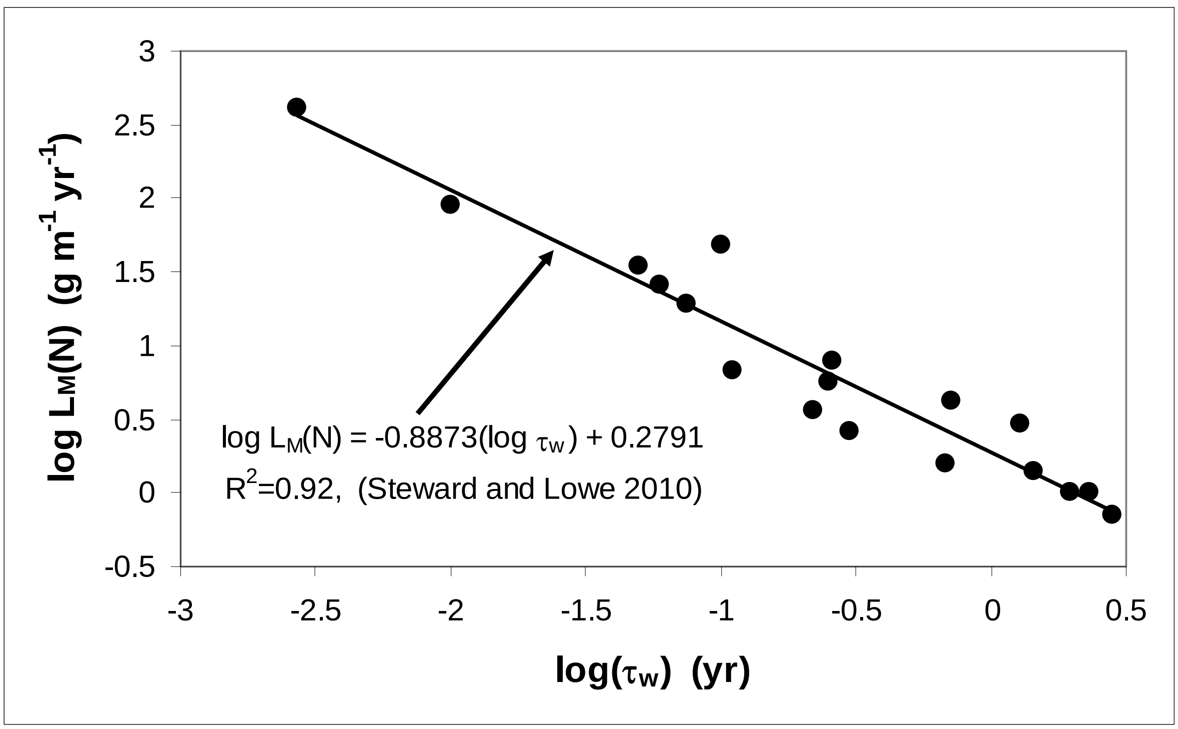

Figure 3-4. Base-10 logarithms of TN load limits established for 18 Florida lakes, river segments, bays and lagoons vs. water residence time (R63 for lakes and river segments, R99 for bays and estuaries). The line is the regression (Equation 3-3) established by Steward and Lowe (2010).

3.3.3 Water Quality Simulation Models

EPA is considering using water quality simulation models as tools for developing numeric criteria for estuaries. Specifically, EPA is considering using hydrodynamic models coupled to water quality models to simulate coupled physical, chemical, and biological processes in Florida estuaries. EPA is considering development of water quality simulation models (i.e, mechanistic watershed models) to provide daily estimates of freshwater and nutrient loading to estuaries as inputs to hydrodynamic-water quality models. Coupled watershed-hydrodynamic-water quality models are widely accepted and have been utilized previously for water quality management purposes. Where EPA utilized simulation modeling, EPA is considering coupling water quality models to state-of-the-art hydrodynamic models because of the often close coupling between water quality processes and physical transport processes in estuaries. The most well-known hydrodynamic-water quality model application in Florida estuaries is the model of the St. Johns River and its watershed (Tillman et al. 2004; Magley and Joyner 2008).

Although water quality models are fundamentally different from regression models, the conceptual approach that EPA is considering is very similar to the approach EPA is considering for use with regression models. Specifically, assessment endpoints and associated biological indicator variables could be used to determine water quality endpoints that can be predicted by the water quality model. The water quality model can then be used to determine TN and TP loading levels or other water quality conditions (e.g., average chlorophyll *a*, estuarine TN and TP concentrations) necessary to ensure water quality that supports balanced natural populations of aquatic flora and fauna. As an example, a coupled watershed-hydrodynamics-water quality model could be used to simulate the impact of TN and TP loading rates on chlorophyll *a*, water clarity, and therefore support for seagrasses growing to an expected depth of colonization. Water quality models could also be used to predict dissolved oxygen concentrations. Criteria, including magnitude, frequency and duration, can be derived for the nutrient parameter simulated by the model that results in attainment of the quantitative endpoint.

The process that EPA is considering for development of numeric criteria using water quality simulation models would involve estimating the current conditions, characterizing natural conditions, and finally developing numeric criteria to translate Florida's existing narrative nutrient criterion. Simulations of observed or "current" water quality conditions are necessary to calibrate the watershed and estuarine water quality models. Typically, data from one or more years could be used to calibrate the water quality models, and data from one or more different years would be used to evaluate the performance of the model. In the case when aquatic life uses are impaired under existing water quality conditions, "natural conditions" would be developed to estimate the TN and TP loading rates and associated water quality responses that could be expected to occur in the absence of anthropogenic disturbance. To characterize natural conditions, the watershed model would be run with all anthropogenic sources removed to determine the concentrations of nutrients, absent any human disturbance. This includes returning all land uses to a natural condition and removing any point sources of nitrogen/phosphorus pollution. The resulting TN and TP loading rates would then be utilized within the hydrodynamic water quality model to simulate the water quality conditions that would be expected to occur in the estuary if TN and TP loading were returned to background levels. Different numeric criteria would be evaluated to determine the highest loading rates that can occur while maintaining simulated water quality conditions that will support the assessment endpoints that EPA has identified. Because simulation models can provide spatially and temporally-resolved outputs, simulated water quality under compliance scenarios could be used to compute spatially-resolved (i.e., estuary segment-specific) estimates for criteria magnitude, frequency and duration.

To consider which models could be used to develop numeric criteria, EPA developed an inventory of the watershed and estuary models that have been previously applied to estuaries in Florida (Table 3-3). EPA's inventory was based on an inventory of models developed by FDEP (Wolfe 2007), with additions based on discussions with FDEP staff. EPA reviewed the models that have been utilized in Florida (Appendix C). Based on the results of its review, EPA is considering using the Loading Simulation Program in C++ (LSPC) for simulating freshwater flows and nutrient loading from watersheds, the Environmental Fluid Dynamics Code (EFDC) for simulation of estuarine hydrodynamics, and Water Quality Analysis Simulation Program (WASP) for simulation of estuarine water quality.

Table 3-3. An inventory of models that have been previously applied to Florida estuaries and their watersheds. The inventory is based upon an inventory compiled by Wolfe (2007) and was developed further based on discussions with FDEP staff. "fecal only" indicates that although a water quality model was developed, it was only used to simulate concentrations of fecal coliform bacteria.

Estuary	Models Used[a]	Watershed Model	Hydrodynamic Model	Water Quality Model
Perdido Bay	EFDC, HSPF	Y	Y	Y
Pensacola Bay				
Escambia Bay	EFDC	N	Y	Y (fecal only)
Escambia Bay	RCA	N	Y	Y
Choctawhatchee Bay		N	Y	N
St Andrews Bay	LSPC, EFDC, WASP	Y	Y	Y
Apalachicola Bay	EFDC, WASP	N	Y	Y (fecal only)
Apalachicola Bay	POM	N	Y	N
Apalachee Bay				
Suwannee River Estuary	EFDC, CH3D	N	Y	Y
Fenholloway River	EFDC, WASP	N	Y	Y
Fenholloway River	RCA	N	Y	Y
Tampa Bay	WAMView, HSPF, WASP, ECOM-3D, CH3D	Y	N	Y
Sarasota Bay	DYNHYD, WASP, CH3D	N	Y	Y
Charlotte Harbor	LSPC, EFDC, WASP, CH3D	Y	Y	Y
Caloosahatchee River Estuary	CH3D	N	Y	Y
Caloosahatchee River Estuary	HSPF, EFDC	Y	Y	Y
Myakka River Estuary	WAMView	Y	N	N
Naples and Rookery Bay		N	Y	N
North Ten Thousand Islands				
South Ten Thousand Islands				
Biscayne Bay	CH3D	N	Y	Y
Indian River Lagoon	CH3D	N	Y	Y
Indian River Lagoon	PLSM, HSPF, regression approach	Y	N	Y
Indian River Lagoon	HSPF, EFDC	Y	Y	Y
Loxahatchee River Estuary	WAMView, EFDC, WASP	Y	Y	Y
St. Lucie Estuary	EFDC	N	Y	N
St. Johns River Estuary	PLSM, HSPF, WAMView, EFDC, CE-QUAL-ICM	Y	Y	Y
St. Marys River Estuary				
Florida Bay	EFDC, HYCOM, USGS-provided Everglades flows	Y	Y	Y

a. Model abbreviations: CE-QUAL-ICM - three-dimensional eutrophication model; CH3D - Curvilinear-grid Hydrodynamics 3d; DYNHYD - WASP hydrodynamics model; ECOM-3D - Estuarine Coastal Ocean Model; EFDC - Environmental Fluid Dynamics Code; HSPF - Hydrological Simulation Program—Fortran; HYCOM - Hybrid Coordinate Ocean Model; LSPC - Loading Simulation Program in C++; PLSM - Pollutant Load Screening Model; POM - Princeton Ocean Model; RCA - Row Column AESOP; WAMView - Watershed Assessment Model; WASP - Water Quality Analysis Simulation Program

3.3.3.1 Watershed Models

Of the eleven water bodies for which watershed models have been developed, seven of the watershed models were developed using either Hydrological Simulation Program—Fortran (HSPF) or Loading Simulation Program in C++ (LSPC). These models are nearly identical in terms of the algorithms used to simulate water flow and water quality, but differ in their software architecture. LSPC has been updated to relax certain computation limitations associated with HSPF, making it easier to apply it to larger watersheds. Aside from HSPF and LSPC, the Watershed Assessment Model (WAMView) has been previously used most often.

EPA is considering applying LSPC to 19 watersheds in Florida (Figure 3-5). These watersheds encompass all of the watershed areas for Florida's estuaries, except for those in South Florida. For watersheds outside of South Florida, EPA can utilize geospatial data collected by FDEP that describes the location and magnitude of flow from springs, to account for water lost to groundwater.EPA is not considering application of LSPC to South Florida due to the complexity of the Everglades canal systems, the high degree of artificial (i.e., human) control of water flow, and the complex interactions between surface waters, ground water, marshes, and wetlands in this area.

LSPC is a comprehensive data management system and model that is capable of representing water flow, water quality, and pollutant loading from nonpoint and point sources and simulating in-stream processes affecting pollutant transport. LSPC is supported by EPA and has been used by FDEP for TMDLs across Florida for estimating watershed loads. LSPC simulates watersheds as a series of hydrologically connected sub-watersheds. LSPC represents receiving waters as one-dimensional, completely mixed stream reaches or reservoirs.

LSPC is a dynamic watershed model driven by time-variable weather input data. It produces time series of flow and pollutants simulating transport in overland flow, the vadose and saturated zones, and instream components of the system, using an area-weighted or "lumped" methodology. LSPC can simulate loadings from multiple land uses and represent instream processes that affect the fate of nutrients within the stream network. Model documentation is available from the EPA Watershed and Water Quality Modeling Technical Support Center (http://www.epa.gov/athens/wwqtsc/).

Input data for LSPC includes three main categories of information: (1) landscape data, including topography, point source locations, locations and connections among streams, etc.; (2) meteorological data, including precipitation, air temperature, humidity; and (3) land use and pollutant-specific data (land use areas, monitoring data, etc.). The watershed loading component of the model divides all land uses into pervious and impervious segments, which are further grouped by land use and subbasin. Loads from subbasins are routed to receiving waters (representative stream reaches or reservoirs).

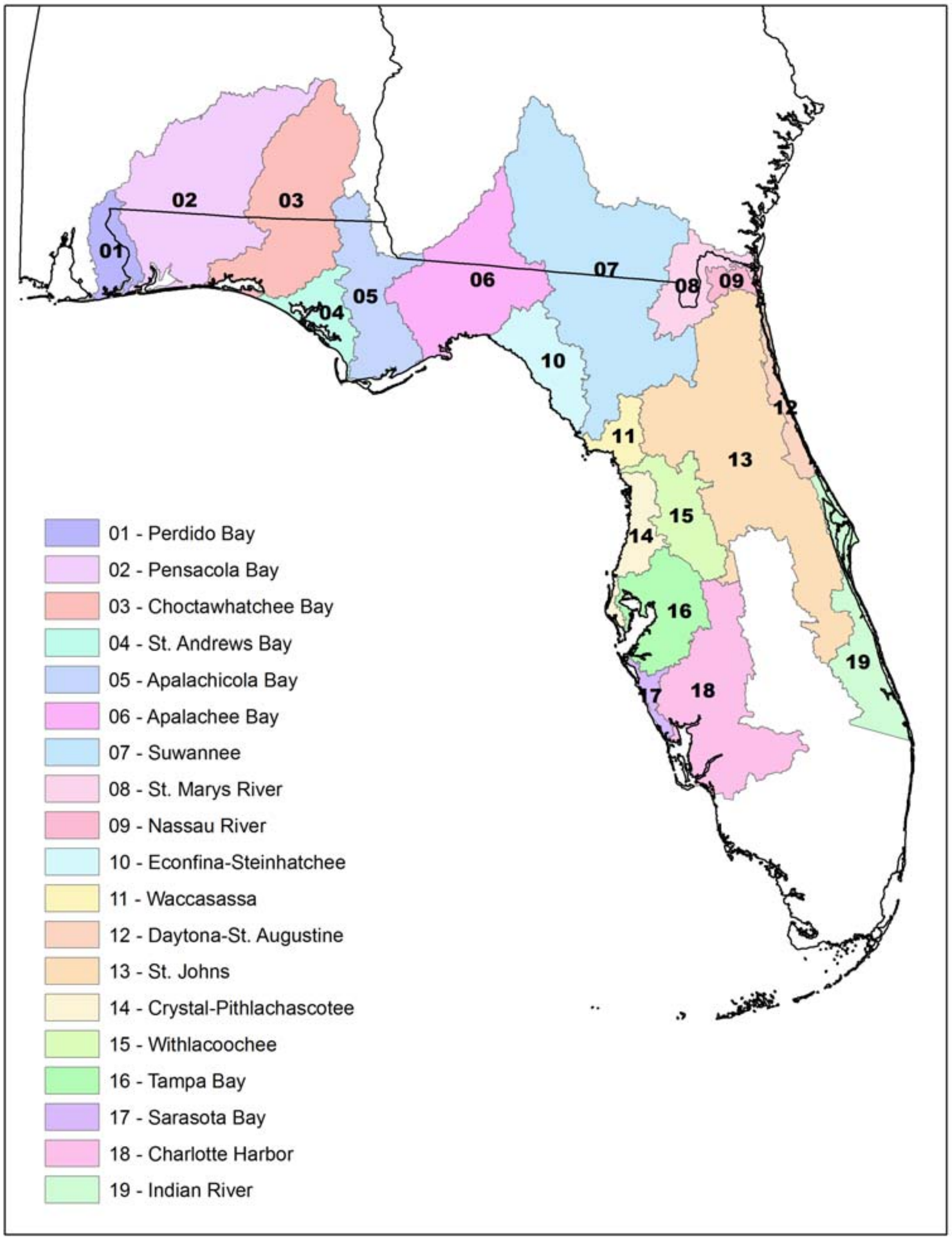

Figure 3-5. Location of 19 drainage areas for which LSPC watershed models are proposed for development

LSPC can evaluate both point and nonpoint sources and simulate both land-based (e.g., rainfall-runoff) and instream water movement and processes. Nitrogen/phosphorus pollution loading may originate from a wide variety of nonpoint sources, such as agriculture, onsite wastewater treatment systems (e.g., septic systems), urban runoff, timber production areas, and atmospheric deposition. Only runoff processes are simulated for impervious land units, whereas interflow and base flows are simulated in addition to runoff for pervious land units.

For each pervious and impervious land segment, LSPC will estimate the water budget, sediment transport, and general water quality constituents, which will represent TN and TP. In each water body segment, LSPC will simulate hydraulic behavior, water temperature, transport of TN, TP, and sediments, and transport of BOD. LSPC simulates nitrogen/phosphorus pollution loading from watersheds using a build-up/washoff approach wherein nutrients accumulate between rain events and are mobilized and transported into streams during rain events. The model utilizes an hourly time step and provides daily average outputs. When using this method, nitrogen, phosphorus and other constituents can be applied to the land surface over time so that a mass of the pollutant accumulates and is subsequently removed at a rate correlated to a corresponding quantity of sheet flow on the land surface.

EPA is considering the use of the following model inputs:

- Subwatershed delineations
  - National Hydrography Dataset (NHD+) catchments
  - Natural Resources Conservation Service (NRCS) 12-Digit HUCs
- Subwatershed information (elevations, slopes, reach lengths, etc.)
  - National Elevation Dataset (NED) 1/3 arc-second (10 meter by 10 meter)
- Land use (2005) provided by FDEP and Water Management Districts
  - Comparable data from Georgia (Georgia Land Use Trends [GLUT] 2005, http://narsal.uga.edu/glut.html) and Alabama (National Land Cover Data [NLCD] 2005, http://www.epa.gov/mrlc/) in applicable watersheds
- Meteorological data
  - National Climatic Data Center (NCDC) rainfall data – approximately 120 rain gauges in Florida (NCDC 2010)
  - Florida State Climatological Office (FL SCO)
- Soils, onsite wastewater treatment systems, and other county-level data
- NPDES-permitted point sources and water withdrawals from Florida's Wastewater Facility Regulation (WAFR) system (http://www.dep.state.fl.us/water/wastewater/facinfo.htm)

Nutrient data from Florida's IWR database and discharge data from USGS flow gauges will provide inventories to select stations to be used in the calibration and evaluation process for hydrology and water quality simulation. Sites with the highest number of observations for parameters such as TN, TP, biological oxygen demand, dissolved oxygen, and water temperature will be selected for calibration. The LSPC model will be run from 1995 through 2009, with 1995 serving as a "spin-up" period (to minimize effects of initial conditions). Model output will include daily outputs for streamflow, average depth, average flow velocity, and TN and TP concentrations for each subbasin of the watershed. Outputs from terminal reaches (i.e., those that discharge to tidal waters) will provide estimates of TN and TP loading rates to estuaries.

3.3.3.2 Hydrodynamic and Water Quality Models

Coupled hydrodynamic-water quality models using the Environmental Fluid Dynamics Code (EFDC) for hydrodynamics and the Water Quality Analysis Simulation Program (WASP) for water quality have been applied to many water quality management projects throughout the Southeast United States and Florida (see Table 3-3). Because of this, EPA is considering using the EFDC and WASP (Version 7.41) models for application to numeric criteria development for Florida estuaries. EFDC and WASP are both publicly available (http://www.epa.gov/athens/wwqtsc/).

3.3.3.2.1 EFDC

The EFDC model is an advanced, three-dimensional surface water modeling system for hydrodynamic and reactive transport simulations of rivers, lakes, reservoirs, wetland systems, estuaries, and the coastal ocean. The modeling system was originally developed at the Virginia Institute of Marine Science as part of a long-term research program to develop operational models for resource management applications in Virginia's estuarine and coastal waters (Hamrick 1992). EFDC is currently used by universities, governmental agencies, and engineering consultants.

EPA is considering application of EFDC to simulate hydrodynamics (i.e., three-dimensional advective transport and mixing) in Florida estuaries. EFDC is run on an orthogonal grid, which will extend from tidal reaches of Florida's streams beyond the ocean and gulf passes, and into coastal waters to ensure that the domain of interest (i.e., estuarine waters) is not substantially affected by artifacts propagating from the seaward open boundary of the model. Inputs to the model include freshwater inflows (from LSPC), precipitation and evaporation, wind speed and direction, solar heating, and tidal forcing at the open boundaries. EFDC will simulate three-dimensional distributions of water temperature and salinity, and will compute advective and diffusive transport in three dimensions. EFDC time-variable outputs will be exported as a hydrodynamic output file, which will be utilized within WASP7 to calculate constituent transports between model elements.

3.3.3.2.2 WASP

The Water Quality Analysis Simulation Program (WASP) is an EPA-developed and supported water quality model that is routinely applied throughout the United States and worldwide to investigate water quality issues. EPA is considering using Version 7.41 (WASP7) to simulate estuarine water quality processes for the purpose of numeric criteria development. WASP version 7.41 is the newest version of WASP (released June 7, 2010) and has many upgrades to the user interface and to the model capabilities. WASP can be downloaded at http://www.epa.gov/athens/wwqtsc/html/wasp.html.

WASP is a dynamic compartment-modeling program for aquatic systems. It can simulate processes in both the water column and underlying benthos. The time-varying processes of advection, dispersion, point and diffuse mass loading and boundary exchange are represented in the basic program. Water quality processes are represented in special kinetic subroutines that are either chosen from a library or written by the user. WASP is structured to permit easy substitution of kinetic subroutines into the overall package to form problem-specific models. WASP comes with two such models—TOXI for toxicants and EUTRO for conventional water quality. EPA's implementation of WASP for criteria development will utilize the EUTRO model.

Earlier versions of WASP EUTRO have been used to examine eutrophication of Tampa Bay; Lake Okeechobee (James et al. 1997); Neuse River and estuary (Wool et al. 2003); the Great Lakes (Thomann et al. 1979; Thomann et al. 1976; Di Toro and Connolly 1980; Thomann 1975), and the Potomac Estuary (Thomann and Fitzpatrick 1982), among many others. In addition to these, numerous applications are listed in Di Toro et al. (1983). Other applications of the WASP in Florida include St. Andrews Bay, Apalachicola Bay, Fenholloway River, Sarasota Bay, Charlotte Harbor, and Loxahatchee River Estuary (see Table 3-3).

Many models, including WASP, include code to enable a wide variety of simulations. The choice of which features should be activated in any particular model implementation will depend upon the target processes, or objectives, of the simulations, which are determined by the questions to be addressed. A complex model that explicitly simulates many processes is not necessarily better than a simpler model that focuses on the aspects of the ecosystem that are most pertinent to the target process (Arhonditsis and Brett 2004). EPA is considering implementations of WASP that are guided by the principle that model functions should be activated only if it is determined to be necessary to achieve model simulation and performance objectives.

For the Pensacola Bay application described below, the EFDC hydrodynamic model was used to perform hydrodynamic simulations, which provide the following information to WASP via a hydrodynamic linkage (HYD) file: (1) time variable exchanges with the Gulf of Mexico (e.g., due to tides), (2) three-dimensional model cell structure and volumes, (3) three-dimensional, time-variable exchanges among model cells (includes horizontal and vertical advective and non-advective exchanges, and (4) three-dimensional, time-variable salinity and temperature. State variables, water quality processes, and boundary conditions for the Pensacola WASP model are shown in Table 3-4. Rates and kinetics were established through parameter calibration techniques to match water quality observations from Pensacola Bay.

Three organic carbon variables play an equivalent role as BOD, representing organic matter that is relatively refractive, of an intermediate reactivity, or labile. Nitrogen is divided into organic and inorganic fractions. Organic nitrogen state variables are dissolved organic nitrogen, labile particulate organic nitrogen, and refractory particulate organic nitrogen. Inorganic nitrogen forms are ammonia and nitrate (nitrite is implicitly represented). Both NO_3^- and NH_4^+ are utilized to satisfy algal N requirements, with NH_4^+ being preferred. The primary reason for distinguishing the two is that ammonia is oxidized by nitrifying bacteria into nitrate. Nitrification can be a significant sink of oxygen in the water column. Sediment nitrification is represented implicitly via the boundary condition value for sediment oxygen demand. As with carbon and nitrogen, organic phosphorus is considered in three states: dissolved organic phosphorus, labile particulate phosphorus, and refractory particulate phosphorus. Only a single inorganic form, orthophosphate, is considered. Orthophosphate exists as several states within the model ecosystem: dissolved phosphate, phosphate adsorbed to inorganic solids, and phosphate incorporated in algal cells. Equilibrium partition coefficients are used to distribute the total among the three states.

Table 3-4. The state variables included in the implementation of WASP7 for Penscaola Bay. In addition, the key biogeochemical processes represented in the model and the boundary conditions that must be specified to run the model.

<table>
<tr><th>State Variables</th><th>Processes</th></tr>
<tr><td rowspan="3">• Ammonia (NH_4^+)
• Nitrate+nitrite (NO_3- + NO_2^-)
• Inorganic phosphorus (PO_4^{3-})
• Dissolved organic nitrogen
• Dissolved organic phosphorus
• Phytoplankton (carbon and chlorophyll *a*)
• Particulate detritus (C, N, and P)
• Carbonaceous biological oxygen demand 1 (CBOD1)
• Nitrogenous biochemical oxygen demand
• Dissolved oxygen
• Total suspended solids</td><td>• Phytoplankton production and mortality
• Light and nutrient limitation (N, P, not Si)
• N and P uptake by phytoplankton
• Mineral sediment sinking (but not resuspension)
• Particulate organic matter sinking
• N and P remineralization
• Nitrification
• Denitrification
• Chemical-biological oxygen demand (3 classes)
• Air-sea exchange of dissolved oxygen</td></tr>
<tr><th>Boundary Conditions</th></tr>
<tr><td>• Sediment oxygen and nutrient fluxes
• Gulf of Mexico concentrations of all state variables
• Inflows of all state variables associated with freshwater inflows and point sources
• NO_3^- concentrations in rainfall</td></tr>
</table>

WASP is capable of simulating four classes of algae, each targeting a specific ecological "niche" defined by distinctive characteristics of the class and the role those characteristics play in ecosystem function. Cyanobacteria, commonly called blue-green algae, are characterized by their abundance as picoplankton in saline water and by their bloom-forming characteristics in fresh water. Being very small, picoplankton do not sink at appreciable rates. Another key feature of cyanobacteria is that some species fix atmospheric nitrogen and can form harmful blooms. Diatoms are distinguished by their need for silica to form their siliceous tests. Diatoms are can grow quickly given sufficient nutrients, but also sink relatively quickly, being large cells and lacking flagella to actively maintain position in the water column. Planktonic algae that do not fall into the two groups are lumped into the heading of green algae. Green algae settle at a rate intermediate between cyanobacteria and diatoms and are subject to greater grazing pressure than are cyanobacteria. The fourth category is macroalgae, which could be expected to increase in biomass when nutrients are available but high flushing rates prevent accumulations of planktonic algae. The algal community of Pensacola Bay has been characterized as dominated by slow-growing non-N-fixing picoplanktonic cyanobacteria during summer and diatoms at other times of the year. Because the cyanobacteria lack a distinctive functional role (i.e., they are not N-fixers), and the Pensacola Bay phytoplankton community can be modeled effectively with a single class of algae, there is not sufficient reason to model the community with more than one class.

WASP is able to simulate sediment-water oxygen and nutrient exchanges by simulating sediment processes using a sediment diagenesis model. This approach entails substantial data requirements, as well as a need for adequate data to calibrate the model. The approach that EPA is considering using for Pensacola Bay is to specify these oxygen and nutrients fluxes as boundary conditions. This may improve model performance relative to a poorly specified and calibrated simulation of these processes. A limitation of the approach is that if changes in benthic processes such as sediment oxygen demand or nutrient fluxes are an important aspect of the response of the ecosystem to changes in nutrient loading, then model will not be able to reflect those responses. For example, if estimates of SOD are based on recent observations (e.g., Murrell et al. 2009), and SOD would decrease if nitrogen/phosphorus pollution loading rates decreased, the model would tend to underestimate the improvement in bottom water oxygen condition that could be associated with reduced nitrogen/phosphorus pollution loading. EPA could address this by modifying the boundary condition.

An important development within WASP7 for numeric criteria development in Florida is the addition of light extinction algorithms, which have been added to allow the model to predict light extinction as a function of background light extinction, attenuation by phytoplankton, and attenuation by suspended solids (including detritus). These new algorithms allow the model to predict changes in light extinction as function of management practices that reduce loading of nutrients and suspended solids. The light model in WASP can be used to determine the availability of light at the bottom, a key indicator of the likely effect of water quality on seagrass growth and survival.

3.3.4 Evaluating Water Quality Simulation Models

Because simulation models are complex computational constructs with many parameters that must be specified, EPA is evaluating systematic and quantitative approaches that the Agency could use to calibrate models, verify their performance against independent data, and evaluate uncertainty associated with model predictions. EPA is evaluating a variety of quantitative performance metrics that have been proposed (e.g., Stow et al. 2009) and is also considering metrics that have been previously applied in regulatory environmental modeling (e.g., Wool et al. 2003).

Model calibration for both watershed models proceeds from physical properties, to chemical properties and ultimately to biological properties and evaluation of specific model outputs that define the model endpoints or objectives. For example, the calibration sequence for LSPC watershed modeling begins with water balance and then proceeds to water temperature and finally water chemistry (e.g., TN and TP). The calibration sequence for EFDC/WASP begins with water levels, salinity and water temperature (EFDC), and proceeds to water quality simulations within WASP, including chlorophyll-a or DO, which are modeling endpoints.

The model evaluation procedure that EPA is considering would involve evaluating models across a range of temporal and spatial scales. For example, hydrologic calibration of watershed models would begin with ensuring the annual water balance, and then would examine seasonal or monthly distributions of discharge, and finally hydrograph components such as base flow and storm flow. EPA is considering application of graphical and quantitative tools such as (1) plots of hourly, daily and monthly time series, (2) monthly scatter plots and balance plots, (3) seasonal plots (i.e., multi-year composites), (4) flow duration curves, (5) flow duration curves, (6) flow accumulation curves, (7) cumulative error statistics, and (8) evaluation of hydrograph components. EPA is considering similar quantitative evaluation procedures to examine stream velocity and water quality (i.e., TN and TP) across a range of time scales and flow conditions. EPA is considering quantitative performance goals based on prior modeling experience to evaluate if the models are performing well (Table 3-5).

EPA is considering evaluating water quality models by considering overall statistical distributions (e.g., cumulative distribution functions), seasonal distributions, and spatial distributions of key water quality variables (i.e., state variables). Indicators of ecosystem function may also be considered and could include process rates (e.g., primary productivity, plankton metabolism) and estimates of important controls on process rates (e.g., measures of light, nitrogen, or phosphorus limitation of algal growth).

Table 3-5. Performance goals for LSPC hydrology simulations, which are based on previous modeling experience. Values are (average of simulated values - average of observed values) / average of observed values. Hydrologic calibrations for longer averaging periods, including the long-term average flow, are expected to be in closer agreement than for shorter periods, especially summer storm volumes, which can depend on rainfall events.

Error Category (Simulated-Observed)	Recommended Criteria (± %)
Total flow volume (i.e., average flow)	10
Lowest 50% of flows	10
Highest 10% of flows	15
Seasonal average flow (spring, summer, fall, winter)	30
Storm volumes	20
Summer storm volumes	50

3.4 Seagrass Depth Targets

As was mentioned in Sections 2.3 and 3.2, EPA is considering using healthy seagrass communities as an assessment endpoint for numeric criteria in Florida estuaries. Seagrasses are present in portions of estuaries throughout Florida, with the exception of estuaries in northeast Florida from the St. Johns River northward to the Georgia border. Seagrasses create valuable habitat and food resources for estuarine biota by providing protection from predators, by providing substrate for production of epiphytic algae and algal grazers. Seagrasses themselves are highly productive and are grazed directly by a variety of species.

Although healthy seagrass communities depend on a variety of physical and biological factors in addition to water clarity (Koch 2001; Heck and Valentine 2007), management efforts to protect seagrass have often focused on improving water clarity because there are clear conceptual and empirical linkages between nutrient loading, water clarity and extent of seagrasses. In short, increased nutrient loading often contributes to increased phytoplankton production and biomass, reduced water clarity, and consequently, degradation and loss of seagrass habitat. Strong relationships have been observed between the maximum depth to which seagrasses grow, referred to as the "depth of colonization," and water clarity (e.g., Duarte 1991; Kenworthy and Fonseca 1996).

Several studies have characterized the depth of colonization of seagrasses in specific Florida estuaries, for example in the Big Bend Region (Iverson and Bittaker 1986); Tampa Bay (Janicki and Wade 1996; Johansson 2002); Indian River (Steward et al. 2005); and Sarasota Bay (Tomasko et al. 2001). However, the depth of seagrass colonization has not been evaluated in detail in many estuaries. Estimates of depth of colonization may be needed for specific segments of estuaries as well as for more than one point in time.

EPA is considering using an approach that involves overlaying geospatial seagrass coverage data on a spatial data set of bathymetric soundings, then examining the fraction of bathymetric soundings that occur within seagrasses as a function of the depth of the soundings (Figure 3-6). EPA has assembled the available seagrass coverages, which includes data spanning 1940 to 2007 (Table 3-6). Seagrasses have has been mapped three or more times in a number of estuaries, generally on the Florida peninsula. Seagrass coverage has been mapped once in several estuaries in the Florida Panhandle and Big Bend Region.

The statistical deepwater edge will be operationally defined to target the middle of the range of depths that occur at the seaward or deepest edge of seagrass beds within an estuary segment (Figure 3-6). An approach similar to that of Steward et al. (2005) may be utilized for a fraction of the data to evaluate the method. Initial estimates of the depth of colonization for each segment may be adjusted from a mean lower low water datum to mean tide level datum using on tide datums published by NOAA for nearby tide stations.

EPA is considering how to best utilize a data set characterizing seagrass colonization depths to determine targets for seagrass colonization depth. Seagrass restoration targets for Tampa Bay were based on the spatial extent of seagrasses that were present circa 1950, less certain areas that due to ongoing dredging or other modifications are not considered recoverable habitat. Depth targets were then determined based on current bathymetric profiles (Janicki and Wade 1996). Similar historical seagrass coverages are available for some estuaries (e.g., St. Andrew Bay, Table 3-6), but are not available for all estuaries in Florida. Steward et al. (2005) present approaches for determining seagrass depth and associated light targets for protection of seagrasses in the Indian River Lagoon. Although the approach they present could not be applied in some parts of Florida due to an insufficient number of coverages, EPA could consider applying elements of the approach, such as utilizing the depth of colonization within reference segments as "upper restoration depths" and the highest value observed for a specific segment as a minimum target for that segment. EPA could consider applying the methods presented here for computing colonization depth to a union seagrass coverage, as suggested by Steward et al. (2005).

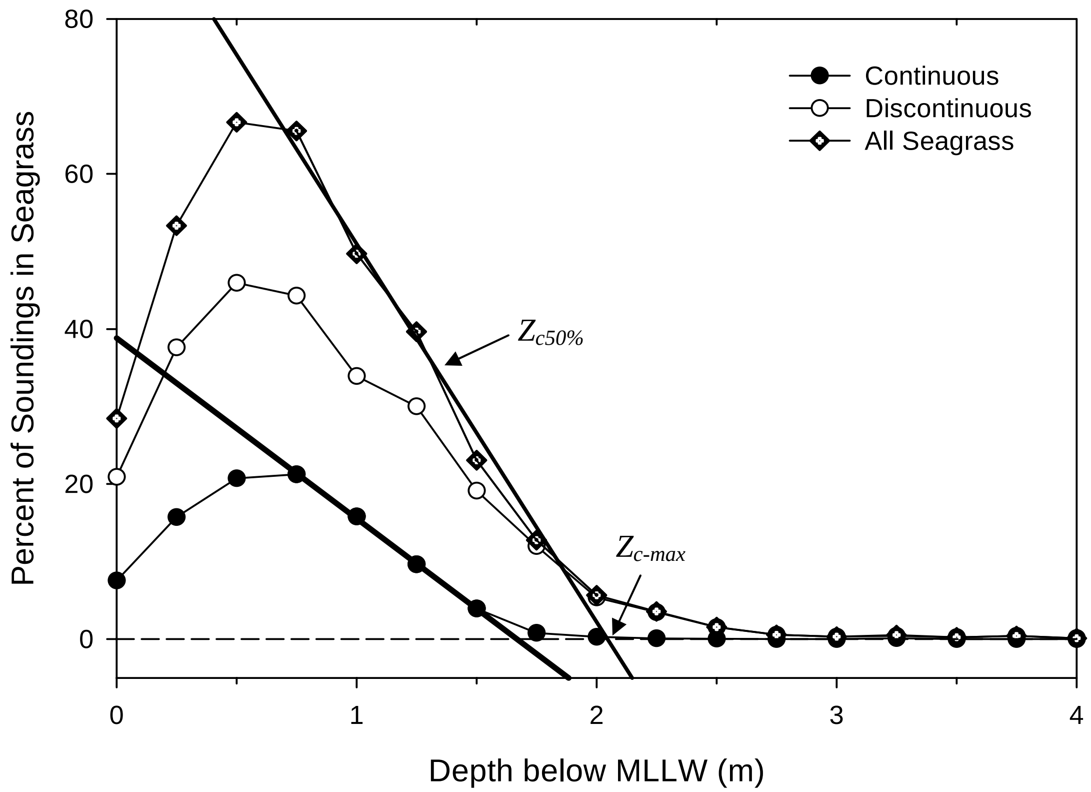

Figure 3-6. An example of the approach used to estimate seagrass depth of colonization, in this case applied to the 1953 seagrass coverage for the East Bay segment (WBID 1061F) of St. Andrews Bay. The estimate of maximum colonization depth (Z_c-max, relative to MLLW) is 1.7 m for continuous seagrass and 2.0 m for all seagrass. The depth at which the percent of soundings in seagrass is decreased to 50% of the maximum is 1.35 m, extrapolated linearly between the observations at 1.25 and 1.50 m.

Table 3-6. Seagrass coverages available for Florida estuaries. CC=complete coverage, PC=partial coverage. USGS = United States Geological Survey, FDEP=Florida Department of Environmental Protection, SWFWMD=Southwest Florida Water Management District, SFWMD=South Florida Water Management District. SJRWMD=St. Johns River Water Management District.

Estuary	Year(s)	Extent	Source
Panhandle Region			
Perdido Bay	1940, 1992	CC	USGS
	1992, 2002, 2003	PC	FDEP
Pensacola Bay	1950	PC	Ollinger et al. (1974)
	1992, 2001	PC	FDEP
	1992	CC	USGS
	2003	CC	FDEP
Choctawhatchee Bay	1992	CC	FDEP
	1992	CC	USGS
St. Andrews Bay	1953, 1964, 1980, 1992	CC	USGS
	1992, 2003	CC	FDEP

Estuary	Year(s)	Extent	Source
St. Joseph Bay – Apalachicola	1992	CC	USGS
Apalachicola Bay	1992	CC	FDEP
	1992	CC	USGS
Big Bend Region			
Apalachee Bay	1992, 2001	PC	FDEP
	1992	CC	USGS
Suwannee River	2001	CC	FDEP
Cedar Key - Suwannee River	1992	CC	USGS
Steinhatchee-Charlotte	1992	CC	USGS
Peninsula Gulf Coast			
St. Joseph Sound - Tampa	1999, 2001, 2004, 2006	CC	SWFWMD
Clearwater Harbor - Tampa Bay	1999, 2001, 2004, 2006	CC	SWFWMD
Crystal Bay - Tampa Bay	1992	CC	USGS
Anclote Keys - Tampa	1992	CC	USGS
Tampa Bay	1950	CC	Haddad (1989)
	1988, 1990, 1992, 1994, 1996, 1999, 2001, 2004, 2006	CC	SWFWMD
	2006	CC	FDEP
Sarasota Bay	1950	CC	
	1988, 1994, 1996, 1999, 2001 2004, 2006	CC	SWFWMD
	1990	PC	SWFWMD
	2006	CC	FDEP
Charlotte Harbor	1950	CC	
	1982 , 1988, 1992, 1994, 1996, 1999, 2001, 2004, 2006	PC	SWFWMD
	2003, 2004	PC	SFWMD
	2004, 2006	PC	FDEP
Caloosahatchee River - Charlotte Harbor	1999	PC	SFWMD
Lemon Bay – Charlotte	1982	PC	SWFWMD
	1988, 1994, 1996, 1999, 2001, 2004, 2006	CC	SWFWMD
Gasparilla Sound -Charlotte Harbor	1982, 1988, 1992, 1994, 1996, 1999, 2001, 2004, 2006	CC	SWFWMD
Matlacha Pass - Charlotte	1999	CC	SFWMD

Estuary	Year(s)	Extent	Source
Harbor			
Pine Island Sound - Charlotte	1999	CC	SFWMD
Estero Bay	1999, 2003, 2004	CC	SFWMD
San Carlos Bay - Charlotte	1999	CC	SFWMD
South Florida			
Rookery Bay	Unknown	CC	FDEP
Ten Thousand Islands	1987, Unknown	PC	FDEP
Biscayne Bay	(1991,92,95); 2004	PC	FDEP
Atlantic Coast			
Indian River	1943, 1986, 1989, 1992, 1994, 1996, 1999, 2003, 2005, 2007	CC	SJRWMD
	2001, 2006	PC	SJRWMD
	2001	PC	SFWMD
	2003	CC	SFWMD
	2005, 2006	PC	FDEP

3.5 Example Application: Pensacola Bay

This section illustrates how EPA could utilize a coupled watershed-hydrodynamic-water quality model to develop numeric criteria for Pensacola Bay.

System Description

Pensacola Bay[9] is located in northwest Florida in Escambia and Santa Rosa counties. The Pensacola Bay system is a complex of sub-estuaries that includes Escambia Bay, Blackwater Bay, East Bay, Pensacola Bay and Santa Rosa Sound (Figure 3-7). The combined system is medium-sized (370 km^2) and shallow (mean depth ~ 3.0 m) and has been characterized as a partially-stratified, drowned river valley estuary (Schroeder and Wiseman 1999). Tides are diurnal and have low amplitude, ranging from 15 cm to 65 cm. The basin includes three major watersheds, which drain via the Escambia, Blackwater, and Yellow Rivers. The Escambia River discharges into Escambia Bay, whereas the Blackwater and Yellow Rivers discharge into Blackwater Bay. Both bays join Pensacola Bay, which connects to the Gulf of Mexico through the narrow (800 m wide) Pensacola Pass. Small openings also permit exchanges with Big Lagoon through a shallow (3 m), narrow (100 m) channel to the west and Santa Rosa Sound to the East through a 1 km opening. Santa Rosa Sound extends 52 km to the east from Pensacola Bay and ultimately narrows to 75 m width before opening into Choctawhatchee Bay. Because of the small size of these channels, flow through them is expected to be minimal compared to Pensacola Pass.

[9] This site description is excerpted from Hagy and Murrell (2007).

Figure 3-7. Location Map of Pensacola Bay

3.5.1 Selection of Assessment Endpoints

EPA evaluated the status of key biological assessment endpoints (Section 3.2) in Pensacola Bay and found evidence that (1) considerable seagrass loss has occurred in Pensacola Bay, and (2) biotic communities in Pensacola Bay are most likely impacted by low dissolved oxygen. However, EPA did not find evidence that harmful algal blooms or proliferations of macroalgae occur in the Bay.

Historical seagrass coverages (e.g., 1950) reveal that seagrasses were historically abundant throughout Pensacola Bay but that much of the historical coverage, outside of Santa Rosa Sound, has been lost (Hagy et al. 2008). Much of the decrease occurred prior to 1980, during a period characterized by high point source nutrient (especially largely nitrogen) loading (Olinger et al. 1975). As of 2003, most of the remaining seagrass area was in Santa Rosa Sound. Historically, low salinity areas such as river deltas were colonized by *Vallisneria americana*; *Rupppia maritima* has been observed at intermediate salinity, whereas *Thallassia testudinum* remains the dominant species in areas with high salinity, such as Santa Rosa Sound. FDEP has recently begun experiments involving transplantation of seagrasses to Escambia Bay, Pensacola Bay, and Big Lagoon.

Following the approach that EPA is considering, EPA will develop estimates of the current and historical depth of colonization of seagrasses in Pensacola Bay (Section 3.3.3.2.2), and subsequently establish targets for depth of colonization. EPA may consider that supporting seagrasses at their colonization depth requires a range of 20-25 percent of incident light (as estimate that has been applied to *Thallassia*), then compute limits for light attenuation that would be consistent with achieving an average depth of colonization equal to the target value. EPA could consider alternative values for the percentage of light required at the colonization depth, recognizing that published estimates have included both lower and higher values (e.g., Steward et al. 2005).

Hagy and Murrell (2007) documented that severe low dissolved oxygen occurs extensively in bottom water areas of Escambia Bay, upper Pensacola Bay, East Bay and Blackwater Bay, with DO levels being less than 2.0 mg/L on a seasonal basis. Hypoxia is confined to stratified bottom waters, whereas surface waters and unstratified areas are well-oxygenated. Limited data are available that document the status of the benthic infaunal community in Pensacola Bay. These data suggest that biomass and abundance of benthos are reduced in areas that are often affected by low dissolved oxygen. FDEP water quality assessments determined that several segments of Pensacola Bay are impaired for low dissolved oxygen.

3.5.2 Numeric Criteria Development

EPA has developed a coupled watershed-estuarine model for Pensacola Bay using LSPC (Section 3.3.3.1) for watershed modeling and EFDC coupled with WASP for modeling estuarine water quality responses (Section 3.3.3.2). EPA is considering using this modeling system to develop numeric criteria for Pensacola Bay. LSPC was used to simulate daily freshwater and nutrient inputs to Pensacola Bay for 1996-2009. Outputs from LSPC are used as inputs to the EFDC/WASP model. WASP would then be used to simulate the effect of TN and TP loading on TN and TP concentrations, chlorophyll *a*, water clarity, and dissolved oxygen in Pensacola Bay. Following the approach outlined in Section 3.3.3, EPA would estimate current conditions, characterize natural conditions, and develop numeric criteria to translate Florida's existing narrative nutrient criterion. Estimates of current conditions would be used for model calibration and evaluation. Natural conditions would utilize LSPC to simulate loading rates, with natural variability in flows and load that would be expected in the absence of a significant anthropogenic contribution. If water quality targets cannot be met without reducing TN and/or TP loading to below the natural background, then EPA could consider establishing TN, TP and chlorophyll criteria based on the characterization of natural conditions from LSPC. On the other hand, if water quality targets can be achieved, then EPA would with vary nitrogen/phosphorus pollution loading rates, ultimately developing a numeric value, which would simulate the highest nitrogen/phosphorus pollution loading rate that could occur while maintaining water quality targets in the estuarine receiving water. EPA could then consider development of criteria for TN and TP loading, estuarine TN and TP concentrations, and chlorophyll *a* concentrations based on the simulated water quality under these conditions. Using time series outputs from the model, EPA could apply a variety of approaches for expressing criteria. For example, time series output could be evaluated in the same manner as data from a reference condition. Because the model output is highly spatially resolved, EPA could develop criteria for subsegments of Pensacola Bay by averaging outputs from model grid cells within the subsegment.

3.5.3 Details on the Pensacola Bay Water Quality Model

3.5.3.1 Watershed Model

In this example, the Pensacola Bay watershed model includes 224 subwatersheds (Figure 3-8). The subwatersheds are based on the NHDPlus catchments and flowlines, but catchments are aggregated to approximately the HUC12 scale to reduce the number of watersheds in the simulation. The LSPC model has 12 terminal reaches discharging directly to the estuary. The model will utilize meteorological data from seven locations across the Pensacola Bay watershed. The Pensacola Bay LSPC model will be calibrated for flow at five USGS flow stations (Table 3-7) and eight associated water quality monitoring stations with data obtained from FDEP (IWR Run 40 database) and the Alabama Department of Environmental Management. With the exception of one station, the calibration points are located in the lower watershed, close to Pensacola Bay. Flow calibrations will ensure that the model correctly simulates overall average flow, seasonal patterns in flow, magnitudes of low and high flow events, and event scale flow patterns. Similarly, water quality calibrations will ensure that the model correctly reproduces overall average loading, seasonal patterns of loading, and magnitude of loading across a range of flow magnitudes.

Table 3-7. USGS flow gauges and water quality monitoring sites within the Pensacola Bay watershed

River	Location	Flow Gage	IWR Gage
Yellow River	Milligan, Fl	USGS 02368000	21FLNWFDS271 21FLBFA 33040004
Big Coldwater Creek	Milton, Fl	USGS 02370500	21FLPNS 33030069
Conecuh River	River Falls, AL	USGS 02372422	n/a
Escambia River	Century, FL	USGS 02375500	21FLGW 3549 21FLBFA 33020001
Escambia River	Molino, FL	USGS 02376033	21FLGW 3541 21FLBFA 33020007

3.5.3.1.1 Watershed Hydrology

Initial values for the hydrological parameters would be taken from default data sets from similar studies in the region. During the calibration process, model parameters would be adjusted, based on local soil types and groundwater conditions, within reasonable constraints until an acceptable agreement was achieved between simulated and observed stream flow. Model parameters that would be adjusted include evapotranspiration, infiltration, upper and lower zone storage, groundwater storage, and losses to the groundwater system.

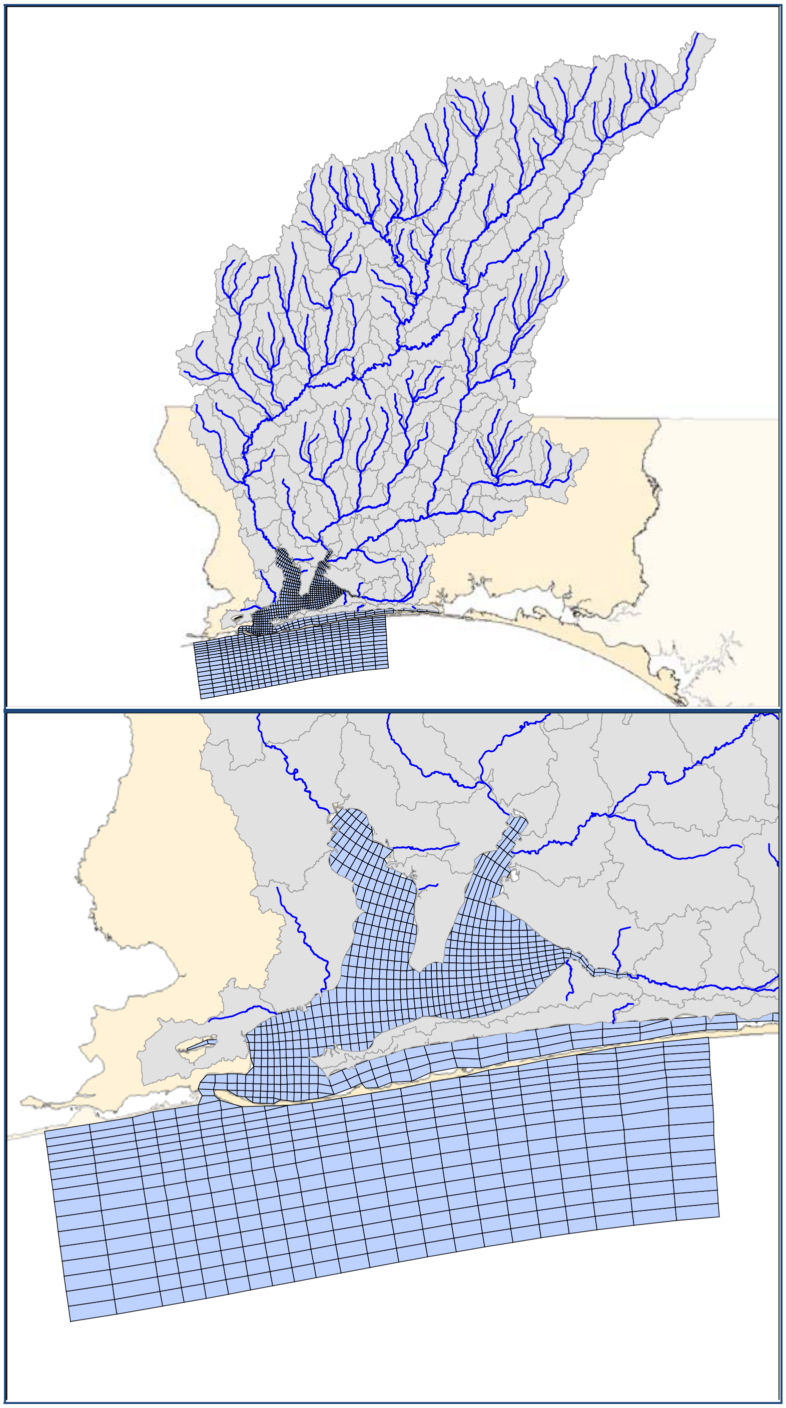

Figure 3-8. (Upper Panel) Map indicating the 224 subwatersheds and associated flowlines that comprise the Pensacola Bay LSPC watershed model. (Lower Panel) The model grid that will be utilized for the EFDC/WASP water quality model for Pensacola Bay.

Historical and short-term USGS flow stations located in the Pensacola watershed would be used to calibrate and evaluate the LSPC watershed hydrology model. There are a total of four USGS flow stations in the Pensacola watershed that have an overlapping period of record with the model simulation. The calibration of the hydrologic parameters would extend from January 1, 1997, through December 31, 2009. Velocity would also be calibrated by adjusting the instream Manning's n value. Simulated velocity would be compared to USGS measured velocities that were taken at the flow calibration stations.

The model would be evaluated by applying the above calibrated hydrologic parameters to other subwatersheds of the Pensacola Bay watershed and comparing the simulated flow to measured flow from a USGS stream gauging station for the same period of time. Care is taken to select watersheds that represent a wide variety of land uses as well as drainage areas. This will help to ensure that the hydrological parameters that were calibrated apply to a wide range of conditions.

3.5.3.1.2 Watershed Water Quality

Following hydrologic model calibration, water quality will be simulated for the watershed. The watershed water quality model would include all point source discharges with permitted flow above 0.1 MGD and nonpoint source contributions. Many components of the water quality model would be established during hydrology modeling, including watershed segmentation, meteorological data, land use representation, soils, reach characteristics, and point source discharges. For application, herein, the water quality model was setup to model water temperature, dissolved oxygen, biochemical oxygen demand, TN, TP, and total suspended solids. EPA is considering modeling TN and TP transport from land to streams using the pervious quality constituent module (PQUAL) of LSPC and instream processes using the generalized quality constituent module (GQUAL).

Reach Group. For the water quality simulations, stream reach processes would be assigned by reach group. Assigning reaches into groups allows for the assignment of unique values, for each reach group, for certain model parameters. The parameters that can be assigned differently by reach group include: sediment bed storage parameters, cohesive and non-cohesive suspended sediment variables for in-stream transport, temperature for stream groups, land to stream mapping, variables associated with BOD sinking, decay, and benthic release, variables for oxygen re-aeration, benthic oxygen demand, oxygen scour, all nutrient parameters, and all plankton parameters. In preliminary analyses for temperature, it was observed that headwater reaches responded differently than non-headwater reaches. Therefore, headwater reaches were assigned to their own reach group and defined as first order streams.

Water Temperature. Soil temperature and heat exchange/water temperature modules with LSPC would be used to simulate water temperature. Simulation of soil temperatures is accomplished by using three layers. The surface layer is the portion of the land segment that determines the overland flow water temperature. The upper subsurface layer determines interflow temperature while the groundwater subsurface layer determines groundwater temperature. Surface and upper subsurface layer temperatures would be estimated by applying a regression equation as a function of measured air temperature and the groundwater subsurface temperature. Soil temperature is used to determine the water temperature of the three different flow paths (surface outflow, upper subsurface/interflow outflow, lower subsurface/groundwater outflow) contributing to stream flow. Once the water is in the stream, the temperature is impacted by mechanisms that can increase or decrease the heat content of the water. Mechanisms which can increase the heat content of the water are absorption of solar radiation, absorption of longwave radiation, and conduction-convection. Mechanisms which decrease the heat content are emission of longwave radiation, conduction-convection and evaporation (Bicknell et al. 2004).

Dissolved Oxygen. LSPC modules for pervious water temperature, dissolved gas concentration, and primary DO and BOD balances would be used to simulate dissolved oxygen. Aside from in-stream transformations, which either consume or produce dissolved oxygen, the primary factors regulating stream DO concentration are stream temperature and atmospheric reaeration. Atmospheric reaeration takes into consideration the initial dissolved oxygen concentration, oxygen saturation level for a given water temperature, water depth, water velocity, circulation, reaeration rate, and a temperature correction coefficient for surface gas invasion.

Sediment. Production and removal of sediment, accumulation and removal of solids and behavior of inorganic sediment modules within LSPC would be used to model sediment loading. EPA would rely on initially establishing detachment coefficients based on similar studies in the region. Depending on the calibration results, this parameter would be adjusted.

Nutrients. LSPC simulates water quality constituents and wash-off of water quality constituents using simple relationships to model nitrogen and phosphorus loadings. Accumulation and wash-off rates play an important role in the determination of nonpoint source loadings to a waterbody. The watershed model must appropriately represent the spatial and temporal variability of hydrological characteristics within a watershed. It must also appropriately represent the rate at which nitrogen and phosphorus components build-up between rain events and wash off during rain events. Key general water quality characteristics include initial storage, wash-off and scour potency, accumulation rates, and maximum storage amounts. The water supplied to a stream from groundwater and through interflow also plays an important role in loading to a waterbody. LSPC allows the user to supply groundwater and interflow concentrations, by hydrologic soil group and land use, by month. The accumulation and wash-off and interflow strongly influence peak flow water quality while groundwater reflects baseflow water quality.

3.5.3.1.3 Development and Calibration

Calibration of the LSPC water quality model includes a sequence of steps that begins with temperature, because the other key parameters are regulated by instream temperature. Temperature would be calibrated by adjusting surface and interflow slopes and intercepts, and groundwater temperature, by land use and hydrologic soil groups until the simulated data closely matched observed. Following temperature calibration, dissolved oxygen would be calibrated with the observed data by adjusting reaeration, and interflow and groundwater dissolved oxygen concentrations.

Finally, calibration would involve BOD, TN, and TP. These three constituents would be modeled by build-up/wash-off and assigned land use-associated concentrations in groundwater and interflow. Build-up/wash-off removes constituents from the land and carries them into the stream. Adjustments would be made to monthly accumulation rate, monthly storage limit, interflow concentration, and groundwater concentration for BOD, TN, and TP until the simulated data was in range with the observed field data.

3.5.3.2 Estuarine Model

As described earlier, EPA proposes to use the EFDC to simulate water surface elevations, currents, salinity and water temperature. The estuarine model grid would include the entire Pensacola Bay system, including Santa Rosa Sound, and will also extend to a seaward boundary approximately 18 km seaward of Pensacola Pass (Figure 3-8). Extending the model well into the Gulf limits the potential for boundary conditions to influence model results in the estuary. The proposed model would resolve 998 grid cells and will have five sigma layers (i.e., it will resolve water column profiles into five vertical layers of equal thickness). The model will be calibrated for the period of 2009 and evaluated for 2002 through 2005. Model calibrations will utilize data collected by EPA approximately monthly at 15 stations during 2002-2004, as well as a continuous record of water quality data recorded in 2009 by an environmental monitoring buoy deployed by EPA. The data include surface and bottom temperature and salinity collected at 15 minute intervals from April to October 2009 at a station in Escambia Bay and May to September 2009 near Navarre Beach. In addition, data on water surface elevations (i.e., tide) collected by the NOAA (Pensacola tidal station 8729840) will be used for calibration. These data include water surface elevation data collected on an hourly basis and daily surface water temperature. Evaluation was performed using data from 2002 through 2004 from 16 stations throughout the estuary with salinity and temperature data.

The output from EFDC would be used to drive the hydrodynamic factors that are used in the WASP model for simulating nitrogen/phosphorus pollution (NO3, ammonia, orthophosphate), phytoplankton (chlorophyll), BOD, DO and total suspended solids. A particularly important feature of WASP is its ability to estimate light attenuation in bottom layers of the estuary. The model would be calibrated with water quality data (nutrients, chlorophyll, and DO) from 2002 from 16 stations throughout the estuary and evaluated using the 2003-2004 data from the same stations. In general, model evaluation ensures that the range of parameter values and seasonal dynamic cycles produced by the model are similar to the observed data.

4 Numeric Nutrient Criteria Development for Florida Coastal Waters

EPA is considering several approaches to derive numeric criteria for Florida's coastal waters. For the purpose of this document, coastal waters are defined as marine waters up to three nautical miles from shore, but exclude estuaries, which are generally semi-enclosed waters.[10] For much of the State's coastal waters, EPA is considering a reference-based approach with satellite remote sensing chlorophyll *a* observations to derive numeric values that translate Florida's narrative criteria and ensure support of a natural balanced population of aquatic flora and fauna.

4.1 Analysis Plan and Delineation of Coastal Areas

At this time, there are variable amounts of monitoring data for Florida's coastal waters. For example, the Northwest Gulf Coast and Atlantic Coast contain little monitoring data of chemical and biological constituents. In contrast, significant monitoring has taken place in coastal waters of South Florida and the Florida Keys as part of the Southeast Environmental Research Center (SERC) water quality monitoring network, and the West Gulf Coast of the Florida peninsula in response to recurring blooms of the toxic dinoflagellate, *Karenia brevis*.

Given the variability of coastal water quality data available in Florida, EPA is considering the use of satellite remote sensing data to derive numeric criteria for coastal waters. EPA's approach is shown in Figure 4-1. First, EPA will delineate the coastal areas, based on the definition of waters of the U.S. in the Clean Water Act and the boundaries of defined estuarine waters. Next, EPA will evaluate satellite-derived chlorophyll *a* (Chl_{RS}-a) to set numeric criteria for chlorophyll as the indicator variable using a reference condition approach in these coastal areas.

EPA recognizes that satellite ocean color remote sensing technology has advanced over the past decade. Chl_{RS}-a in the ocean are a routine measurement and available globally (O'Reilly et al. 1998). Coastal physical forcings such as wind, currents, and tides are known to influence coastal chlorophyll dynamics together with nitrogen/phosphorus pollution loadings from the land. It is expected that all these processes will be represented when using remote sensing as a reference condition approach. Specifically, EPA is considering the use of this method to develop numeric chlorophyll criteria for the Northwest Gulf Coast, West Gulf Coast, and Atlantic Coastal Areas (Figure 4-2).

Due to the increased interference from colored dissolved organic matter (CDOM) and bottom reflectance on satellite measurements, EPA is not considering the derivation of chlorophyll criteria using remote sensing data in coastal waters from Apalachicola Bay to Suwannee River (Big Bend) and South Florida. EPA's approach for developing numeric criteria for these waters is described in Chapter 3 and 5, respectively.

[10] The Clean Water Act's definition of "waters of the U.S." is limited to three nautical miles offshore.

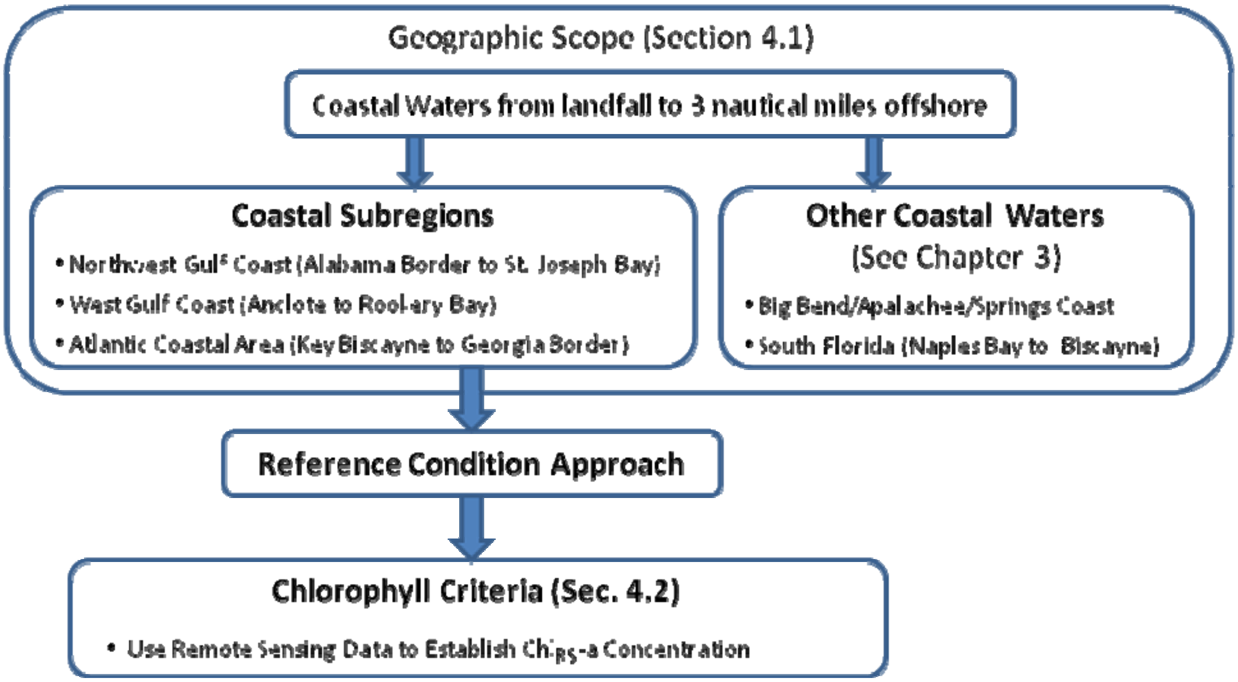

Figure 4-1. Flow chart for the development of numeric nutrient criteria for coastal waters.

EPA is currently not considering the derivation of separate TN and TP criteria for Florida's coastal waters because EPA believes that numeric criteria derived for estuarine waters (see Chapter 3) will inherently protect coastal waters from anthropogenic TN and TP from streams and estuaries. The coastal chlorophyll criteria will be an independent means to assess whether coastal waters are maintaining water quality. Preliminary analysis by EPA indicates that coastal segments adjacent to estuary passes have higher Chl_{RS}-a than coastal segments further from estuary passes. Examples of coastal chlorophyll responding to nitrogen/phosphorus pollution loading and river discharge are included in Walker and Rabalais (2006), where Mississippi River nitrogen/phosphorus pollution loads were empirically related to coastal chlorophyll concentration in the northern Gulf of Mexico. Nababan et al. (*In press*) also reported river discharge was the dominant factor controlling coastal chlorophyll concentration in Mobile Bay, Alabama, and the Florida Panhandle.

Unlike estuaries, which are commonly dominated by freshwater flows and can be organized within watershed boundaries, coastal waters have open offshore boundaries and localized influences near the estuary/coastal boundary (i.e., estuary pass). Numeric criteria for Florida's coastal waters could be derived on a regional scale (i.e. Northwest Gulf Coast, West Gulf Coast, and Atlantic Coast) or at a finer scale. In the latter case, EPA recognizes that FDEP's Water Body Identification Numbers (WBIDs) for coastal waters start at land fall and extend seaward for three nautical miles. Coastal WBIDs are also typically centered at an estuary pass. Thus, EPA will consider the coastal WBID scheme for the basis of organizing defined coastal segments.

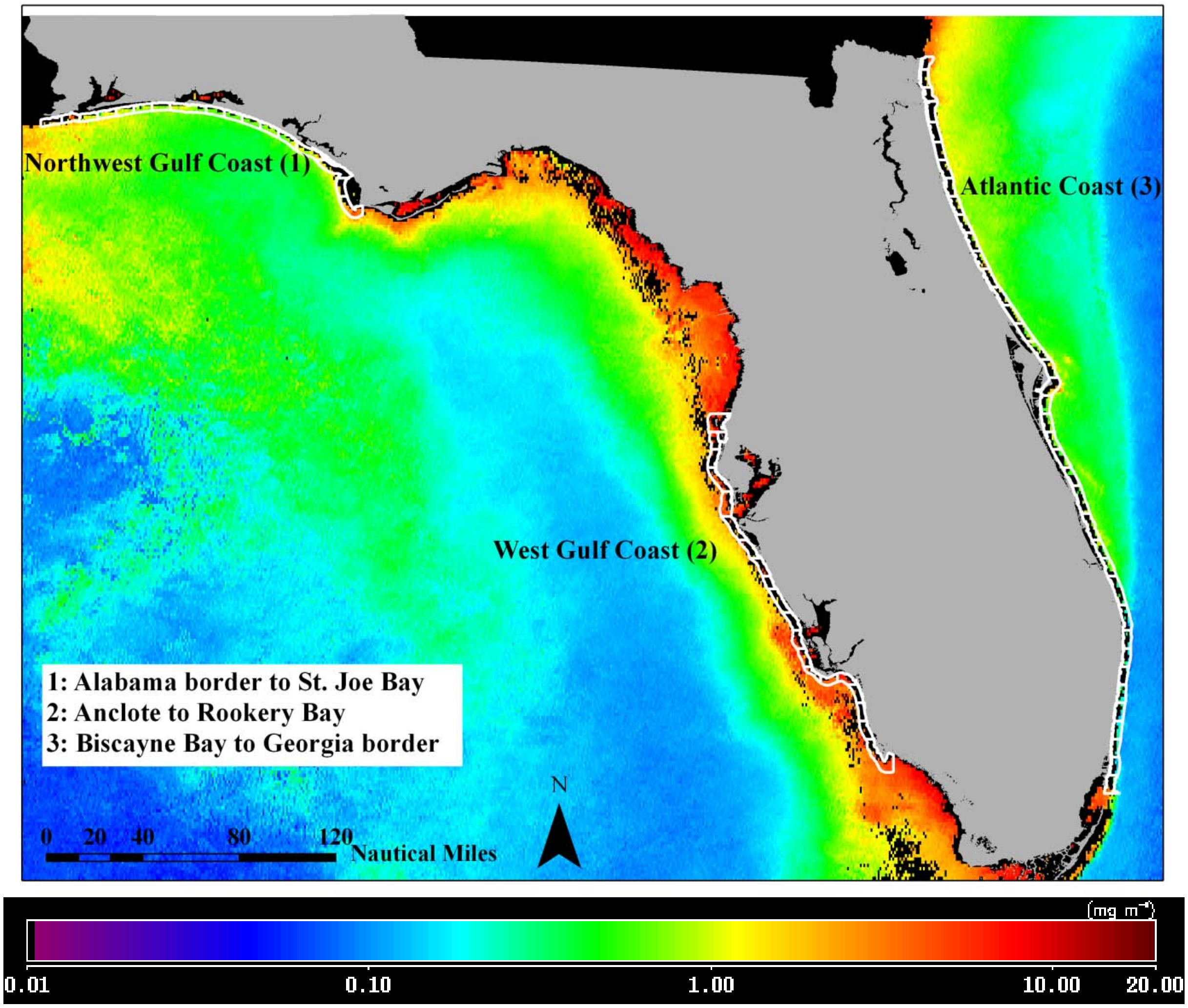

Figure 4-2. Map of coastal segments used in satellite remote sensing analysis. The remote sensing data shown in this figure are a composite result from July 1-August 31, 1999. A remote sensing approach is proposed to develop numeric chlorophyll criteria for the Northwest Gulf Coast, West Gulf Coast, and Atlantic Coastal Area based on establishment of a reference condition using remotely sensed chlorophyll *a* observations (Chl_{RS}-*a)* as the response variable. EPA's approach for developing numeric criteria for South Florida and the coastline from Apalachicola Bay to Suwannee River is described in Chapter 3.

4.1.1 Use of Remote Sensing Data

EPA is evaluating the use of the following satellites to provide measurements of Chl_{RS}-a:

- Sea-viewing Wide Field-of-view Sensor (SeaWiFS)
- Moderate Resolution Imaging Spectroradiometer (MODIS)

SeaWiFS provides a historical time-series of Chl_{RS}-a back to 1997, whereas MODIS data began in 2002. SeaWiFS and MODIS can both provide a continuous temporal measure of Chl_{RS}-a in the Florida coastal waters. Figure 4-3 provides the number of data points that occur within all of Florida's coastal waters from satellite and traditional field samples. Traditional field samples account for 1,648 chlorophyll observations from 1998 to 2010. In contrast, satellites measured

25,549 observations over the same period. When summarized to 8-day averages, a single satellite will typically provide a composite Chl_{RS}-a observation for 40 of the 91 coastal segments each week. Table 4-1 presents data from IWR Run 40 over a 20-year period from 1990 to present for three coastal water bodies.

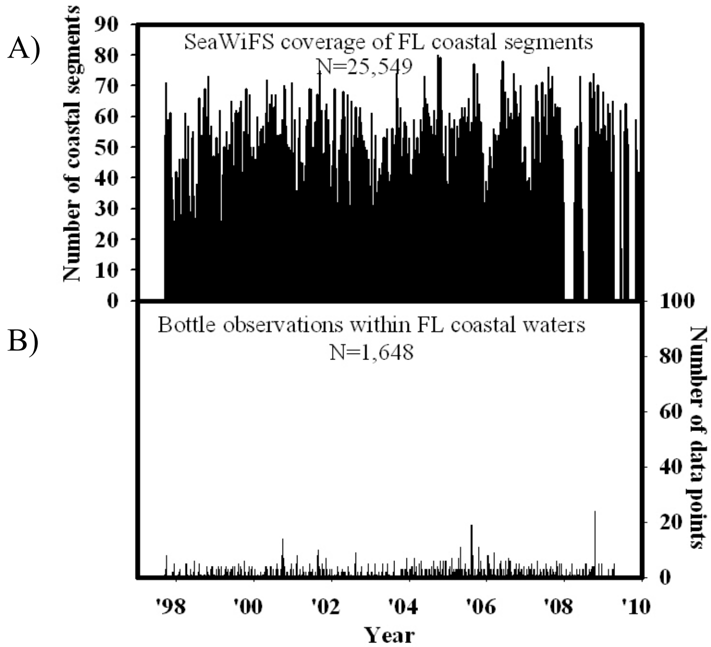

Figure 4-3. (A) Satellite Chl_{RS}-a coverage of the 91 coastal segments between 1998 and 2010. (B) Number of chlorophyll field observations that occur within all Florida coastal waters

Table 4-1. Number of observations in IWR Run 40 since 1990 for chlorophyll, TN and TP for coastal waterbodies

Waterbody	Years	Chl-a, uncorrected	Chl-a	TN	TP
Northwest Gulf Coast	1990–2010	311	241	180	156
West Gulf Coast	1990–2009	865	498	1,862	1,932
Atlantic Coastal Areas	1990–2008	51	16	50	73

The technical advances in satellite imaging and processing has resulted in numerous applications of remote sensing products to address needs for environmental resource management. The following is a brief overview of successful applications of remote sensing toward management decision support.

The NOAA Harmful Algal Bloom Operation Forecast System (HAB-OFS) uses remote sensing as an advanced warning system of harmful algal blooms. This system provides information on the HAB location and the potential for bloom development along the coastal waters of Florida. NOAA Fisheries Service uses remote sensing to characterize sea surface temperature around the Monterey Bay National Marine Sanctuary and delineate the marine reserve boundaries. NOAA Fisheries Service is also utilizing remotely sensed sea surface temperature, sea surface height, and chlorophyll estimates to monitor the recruitment, survival, and productivity of managed fish species, and those protected by the Endangered Species and Marine Mammal Protection Acts.

NOAA is currently using remotely sensed sea surface temperature for beach closure applications within Chesapeake Bay. Two specific projects in Chesapeake Bay are (1) forecasting nuisance jellyfish using remotely sensed sea surface temperature and hydrographic models and (2) mapping the occurrence of bacterial pathogens in the genus *Vibrio*. Satellite applications pertaining to *Vibrio* also extend to the management of oysters in the Gulf of Mexico, where sea surface temperature and *Vibrio* levels are used to inform fisheries managers whether oysters are harvestable or if shellfish bed closures are required.

The SPOT 5 satellite (French Space Agency) provides data that are subsequently classified into geospatial imagery of land cover. Maryland's Department of Natural Resources uses SPOT data to denote tidal from non-tidal wetlands. Legislation in Maryland requires that wetlands be classified, and the State currently uses maps solely generated with SPOT, which are supplied to all coastal managers. South Carolina's Office of Ocean and Coastal Resource Management utilizes LIDAR data to determine Beachfront jurisdictional lines, which have proven to be highly accurate and adhere to laws requiring the mapping of beach setback in the State.

The National Aeronautics and Space Administration (NASA) provided remote sensing observations during the Deepwater Horizon BP oil spill in the Gulf of Mexico at the request of U.S. disaster response agencies. Remote sensing was used by NOAA, the U.S. Geological Survey, and the Department of Homeland Security for monitoring the spill. Finally, EPA used remote sensing to support the Spill Prevention Control and Countermeasures (SPCC), the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA), and the Resource Conservation and Recovery Act (RCRA) (USEPA 2007).

4.2 Deriving Numeric Criteria

EPA is evaluating the use of remote sensing to provide Chl_{RS}-a distributions for the Northwest Gulf Coast, West Gulf Coast, and Atlantic Coastal Area (Figure 4-2). These coastal waters extend along the Northwest Gulf Coast from the Alabama border to St. Joseph Bay; the West Gulf Coast from Anclote to Rookery Bay; and along the Atlantic Coast from the Georgia border to Biscayne Bay. Currently, only WBID 8037 (Waccasassa River, north of the West Gulf Coast) has been identified as impaired for nutrients and chlorophyll in 2005 according to FDEP's CWA section 303(d) listing. Therefore, EPA is considering applying a reference condition approach to derive numeric chlorophyll criteria for Florida's coastal waters to measure the assessment endpoint of balanced phytoplankton biomass and production in order to translate Florida's narrative nutrient criteria into numeric values that support balanced natural populations of aquatic flora and fauna.

Reference condition approaches can take a variety of forms, defined by the source of the reference condition. EPA has previously recommended (e.g., USEPA 2001) that a percentile of water quality measurements in a sample of minimally-impacted waterbodies, which are presumed to be fully supporting designated uses (i.e., not impaired), could serve as criteria for water quality in similar waterbodies. In this case, a reference condition is derived from a reference population of waterbodies. As discussed elsewhere in this document (Chapter 3), EPA considers a reference condition approach to be most applicable when (1) historical data adequately describe water quality conditions when the waterbody was minimally-impacted by nitrogen/phosphorus pollution and was supporting balanced natural populations of aquatic flora and fauna (i.e., historical reference condition) or (2) when the waterbody is currently minimally-impacted by nitrogen/phosphorus pollution and currently supporting balanced natural populations of aquatic flora and fauna (i.e., current-conditions reference condition). EPA has not yet identified any impaired coastal waters for which historical water quality data could provide a suitable reference condition. Therefore, EPA is largely considering using a reference condition approach only in the second case, namely when current water quality conditions are supporting balanced natural populations of aquatic flora and fauna. Further discussion of the reference condition approach applied to coastal waters in Florida is discussed in Section 4.2.3.

4.2.1 Satellite Chlorophyll Data Evaluation

Satellite ocean color data were obtained from NASA Ocean Color Web (Feldman and McClain 2010). SeaWiFS and MODIS satellites provided daily images with 1 km binned spatial resolution when not masked with quality control flags. SeaWiFS and MODIS imagery spatially covered between 31.0 to 23.0^{o} N and 88.0 to 79.0^{o} W. SeaWiFS Data Analysis System (SeaDAS) version 6.1 (Baith et al. 2001) was used to process SeaWiFS and MODIS data from level-1 to level-3 composites and derive Chl_{RS}-a concentrations (NASA n.d.). Results within 1 km of the shoreline were not included in the averaging of Chl_{RS}-a by using quality control flags in the SeaDAS software, and by averaging bins that only fell completely within the coastal segment perimeter line.

Satellite validation of the derived chlorophyll product against field chlorophyll measurements was performed using the native resolution of the sensor. Satellite match-ups were evaluated with a Type II geometric mean linear regression (Laws and Archie 1981) between a 3x3 pixel extraction of satellite data centered at the corresponding in-situ measurement location. Any 3x 3 pixel extractions that had greater than 4 pixels flagged were excluded. Satellite data were filtered for SeaDAS default flags (i.e., cloud contamination, land, and atmospheric correction failure) and any data with viewing angles greater than 60^{o} and solar angles greater than 75^{o}. In-situ data were filtered so only samples collected within ±3 hours of the satellite overpass were utilized (Bailey and Werdell 2006). Field chlorophyll data used for satellite validation were from the following sources:

- Northeastern Gulf of Mexico (NEGOM) project (NOAA National Oceanographic Data Center)
- Ecology and Oceanography of Harmful Algal Blooms project (ECOHAB)
- Fish and Wildlife Research Institute (FWRI)
- Mote Marine Laboratory
- SeaWiFS Bio-optical Archive and Storage System (SeaBASS) (Werdell and Bailey 2002; Werdell et al. 2003).

EPA is considering excluding from the statistical distribution of affected coastal segments any Chl_{RS}-a measurements taken during *Karenia brevis* bloom events. While Vargo (2009a) has concluded that "there is no single hypothesis that can account for blooms," removing data collected during bloom events would ensure that the data points used to derive the statistical distribution of reference conditions would represent conditions that are supporting balanced natural populations of aquatic flora and fauna (See Appendix B for a more complete review of *K.*

brevis as an assessment endpoint). *K. brevis* cell abundance observations from the Florida Fish and Wildlife Conservation Commission FWRI in St. Petersburg can be used to remove the chlorophyll signature that resulted from the onset of *K. brevis* blooms (Figure 4-4). Satellites detect *K. brevis* blooms when cell counts are above 50,000 cells/L (Heil and Steidinger 2009). Coastal segments with an FWRI count greater than 50,000 cells/L during an 8-day composite would be flagged and not averaged into the Chl_{RS}-a distributions. In addition, the same WBID will be flagged one week prior and after a bloom detection to provide a temporal buffer as blooms are transported along the coast.

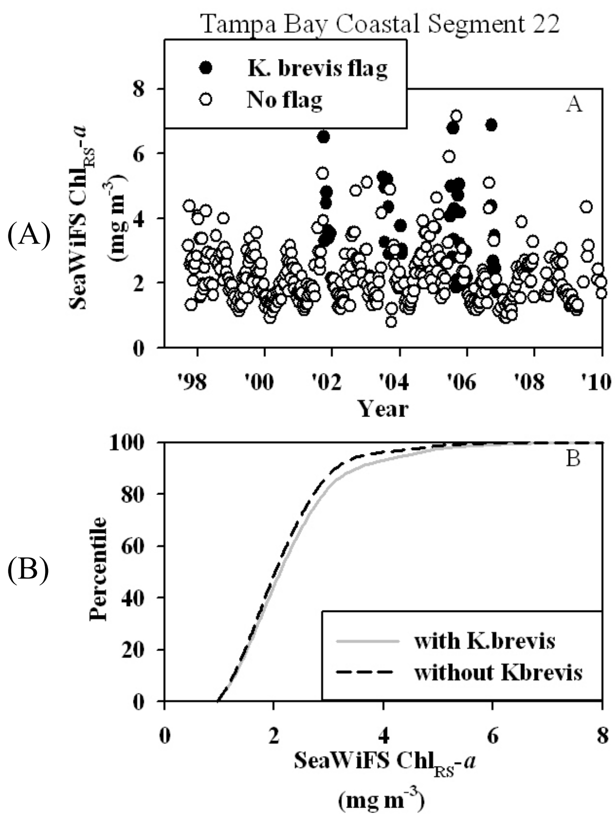

Figure 4-4. Example of (A) *K. brevis* bloom occurrence at the Tampa Bay coastal segment. Black points are included in the Chl_{RS}-a distribution. Red points indicate Chl_{RS}-a values that would be excluded from the distribution because a *K.brevis* count of 50,000 cells/L or higher was measured. (B) Cumulative distribution of Chl_{RS}-a at the Tampa Bay coastal segment with and without *K. brevis*.

4.2.2 Comparison of Satellite Chlorophyll to Field Collected Chlorophyll

There are limited field-collected water quality indicator variable data from throughout the coastal waters of Florida. However, the available field data that coincide with Chl_{RS}-a will be assessed and considered in EPA's approach. The Chl_{RS}-a data was paired with 62 field observations within the three nautical mile boundary to provide a calibration of Chl_{RS}-a data (Figure 4-5). There was a significant positive correlation between field observations and SeaWiFS Chl_{RS}-a (slope = 0.85, $R^2 = 0.52$, $p < 0.0001$). Therefore, the slope can be applied to the Chl_{RS}-a as a correction factor.

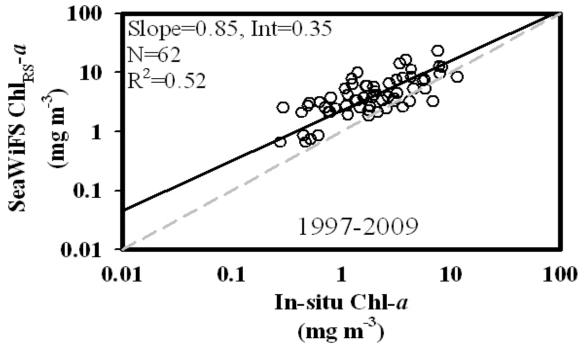

Figure 4-5. Regression of SeaWiFS ChlaRS-a versus field observations from NEGOM, ECOHAB, FWRI, Mote, and SeaBASS which occur within the 3nm coastal boundary (grey dashed line is 1:1 fit and black line is regression slope)

This section compares Chl_{RS}-a for five coastal segments and two monitoring sites near Tampa Bay (Figure 4-6). WBID 8049 is located at the Tampa Bay estuary pass, WBIDs 8047 and 8048 are located further north, and WBIDs 8050 and 8051 are located further south. The two monitoring sites with uncorrected chlorophyll *a* field data are located in WBID 8049. Figure 4-7A displays the time series of Chl_{RS}-a from each of the five WBIDs with two general patterns emerging: (1) Chl_{RS}-a demonstrates seasonal variability with higher concentrations in the late summer and early fall, and (2) Chl_{RS}-a concentrations are generally higher in the coastal segment corresponding to WBID 8049 (located at the Tampa Bay estuary pass) relative to WBIDs 8047, 8048, 8050, and 8051. Figure 4-7B displays the two monitoring sites' uncorrected chlorophyll *a* results in WBID 8049 from 2000-2006. In general, the chlorophyll data from the two sites shown in Figure 4-7B trend similar to that of the Chl_{RS}-a concentrations in Figure 4-7A.

The similarity of the chlorophyll results from remote sensing and field collected data are better represented by displaying the time series data with cumulative sum (CUSUM) charts. As used herein, cumulative sum charts display the cumulative sum of deviations between the result (i.e., chlorophyll result) and its overall mean. The data are transformed by subtracting the mean and dividing by the standard deviation (i.e., computing z-scores) referred to as Z-CUSUM charts. Figure 4-8A is a Z-CUSUM chart for the Chl_{RS}-a from each of the five coastal segments shown in Figure 4-7A. Figure 4-8B is a Z-CUSUM chart of FDEP uncorrected chlorophyll *a* results from two sampling sites located in WBID 8049. In both Figure 4-8A and Figure 4-8B, there is an increase in the Z-CUSUM in 2005 representing greater than average chlorophyll levels. The Z-CUSUM values in the year following 1998 and 2005 (i.e., 1999 and 2006) are approximately the same followed by a decreasing pattern. This indicates that the chlorophyll in the year following 1998 and 2005 is near average, and the chlorophyll in the second year following 1998 and 2005 (i.e., 2000 and 2007) is below average. These results provide additional evidence that Chl_{RS}-a and field collected data provide similar results.

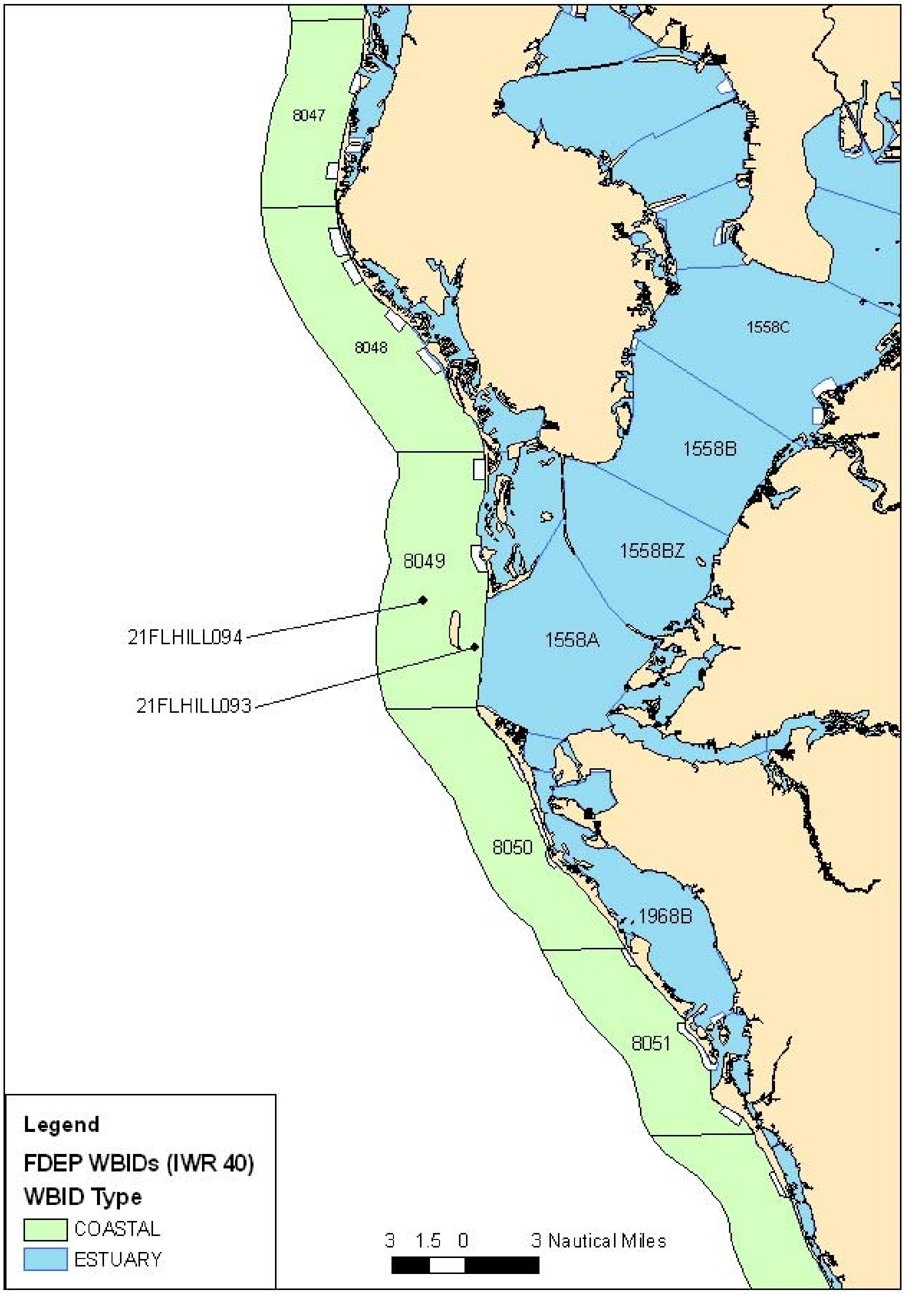

Figure 4-6. Tampa Bay map displaying the location of five coastal WBIDs and two monitoring sites near Tampa Bay. WBID 8049 is located at the Tampa Bay estuary pass, WBIDs 8047 and 8048 are to the north, and WBIDs 8050 and 8051 are to the south. The two monitoring sites with uncorrected Chl-a field data are labeled as 21FLHILL094 and 21FLHILL093.

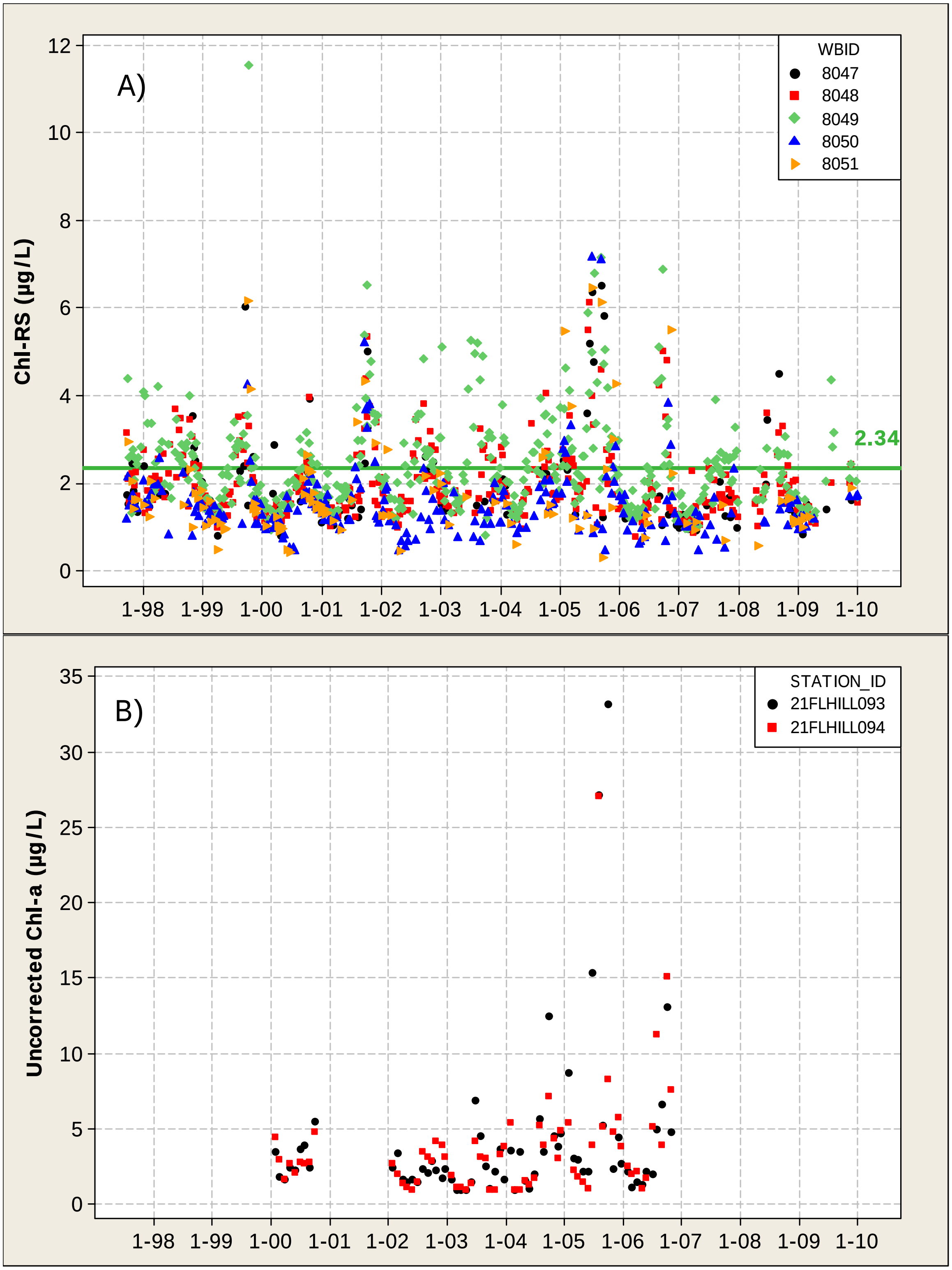

Figure 4-7. (A) Chl_{RS}-a (µg/L) for five coastal segments near Tampa Bay. The overall mean Chl_{RS}-a for the coastal segment corresponding to WBID 8049 is 2.34 µg/L. (B) IWR40 field Chl-a (uncorrected) from two stations in WBID 8049 (note that there are no field data between 1-01 and 1-02)

4.2.3 Chl_{RS}-a Criteria from Statistical Distributions

EPA is considering the use of the field adjusted Chl_{RS}-a to establish chlorophyll distributions (Figure 4-9) for each coastal segment within the Northwest Gulf Coast, West Gulf Coast, and Atlantic Coast regions. EPA is considering removing influences from *K. brevis* blooms from the field adjusted Chl_{RS}-a distribution then determining the numeric criteria value from an appropriate percentile of the distribution. EPA is considering two approaches for deriving criteria using a reference condition approach from a statistical distribution. One approach is to estimate the annual geometric mean concentration and the second approach is a binomial test to maintain current conditions.

4.2.3.1 *Chl_{RS}-a Annual Geometric Mean Concentration*

The geometric mean is preferred to the arithmetic mean because the geometric mean is a better estimator of the central tendency when the data are log-normally distributed as commonly the case for chlorophyll. One approach to compute the annual geometric mean from a data set is to transform the concentrations to a natural logarithm, and then compute the mean by coastal segment and year. The 75^{th} percentile annual geometric mean would be computed as

$$GM_{75} = e^{(\bar{x}+0.6745s)}$$

where 0.6745 is the inverse of the standard normal cumulative distribution with a probability of 0.75 and s is the corresponding standard deviation.

It has been recommended that criteria frequency ranges from one in three years to two in five years would protect designated uses. The GM_{75} would be exceeded, on average, 25 percent of the time without any water quality changes. If the annual geometric mean exceeded the GM_{75} in two of three years, there is 84 percent confidence the annual geometric mean of the test data exceeded the GM_{75}. If the annual geometric mean exceeded the GM_{75} in three of five years, there is 90 percent confidence the annual geometric mean of the test data exceeded GM_{75}.

Figure 4-8. Z-CUSUM plot for (A) Chl_{RS}-a (µg/L) for five coastal segments near Tampa Bay and (B) IWR40 field Chl-a (uncorrected) from two stations in WBID 8049 (note that there are no field data between 1-01 and 1-02 [see Figure 4-7B]. Therefore, the straight line in 4-8B, between 1-01 and 1-02, is an artifact of data availability.)

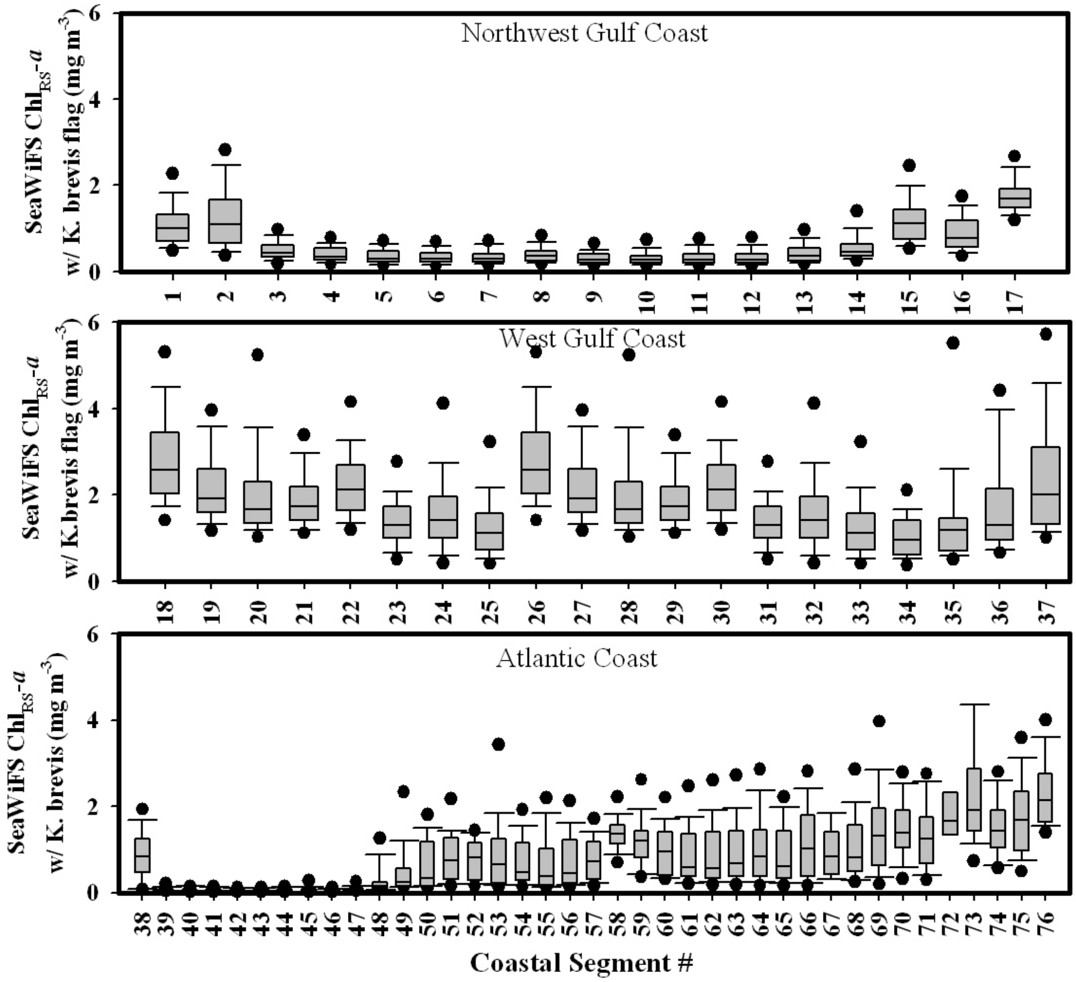

Figure 4-9. Field-adjusted Chl_{RS}-a (µg/L) distributions for all coastal segments between 1997 and 2009 with representations of the minimum, 10^{th}, 25^{th}, 50^{th}, 75^{th}, 90^{th} and maximum values. Northwest Gulf Coast segment 1 is located at the Alabama/Florida border and segment 17 is at St. Joseph Bay. West Gulf Coast coastal segment 18 is located at Anclote Bay and segment 38 is at Rookery Bay. Atlantic Coast segment 39 is located at northern Biscayne Bay and segment 77 is at the Florida/Georgia border.

4.2.3.2 Chl_{RS}-a Binomial Test

Another method that EPA is considering is to specify numeric criteria applicable to individual observations rather than annual statistics. For example, short duration increases in chlorophyll are expected, but these events might not be represented in annual means.

EPA is considering a binomial threshold to set $C_{0.50}$ and $C_{0.25}$ to the maximum 50^{th} and 75^{th} percentile concentration for any 3-year period from the supporting data set. This approach (1) provides a quantitative evaluation of the probability that the data collected during the assessment period are from the same or nearly the same distribution as the observations during the baseline period; (2) is resistant to bias by high values; (3) does not require or imply an assumption of normality or log normality; (4) is minimally affected by censored data; and

(5) tests for changes in both the central tendency and the upper end of the concentration distribution.

4.3 Uncertainties and Data Gaps

Werdell et al. (2009) showed SeaWiFS and MODIS could successfully be used to evaluate chlorophyll in coastal environments such as the Chesapeake Bay. CDOM and bottom reflectance are typically cited as possible interferences with satellite measurements in coastal waters. However, bottom reflectance can be minimized by incorporating the stray light contamination flag. This flag excludes any near shore bins where reflected light from land enters the satellites field of view. Seagrass may provide interferences with satellite derived chlorophyll though seagrasses are not present in coastal waters of the Northwest Gulf Coast, Atlantic Coast, and most of the West Gulf Coast. The northern West Gulf Coast between Cedar Key and Anclote Bay and southwest Florida between Gullivan Bay and Florida Bay have seagrass coverages that occur in coastal segments (Carlson and Madley 2006). However, satellite imagery is not used for coastal criteria in these seagrass coverage locations.

Satellites do not provide direct measurements of chlorophyll as do bottle samples, which are commonly considered acceptable across many laboratories and agencies. However, the data presented in Section 4.2.2 provide strong evidence that remotely sensed chlorophyll is empirically related to field observations and thus exhibits similar trends and patterns as field observations. Florida's coastal waters have not been monitored comprehensively, and regular monitoring of coastal areas is challenging. As a consequence, there are limited data based on field-sampling to support numeric criteria development for all of the State's coastal waters.

To improve on the proposed approach, continuous monitoring by satellite would be necessary to derive more refined criteria. A potential concern is that all satellite missions have a finite duration and may not be always available if the State chooses to use satellite data to assess their coastal waters. EPA recognizes next generation satellites and algorithms will be developed in the future, and criteria values will need to be reviewed and adjusted with improving technologies. In addition to SeaWiFS, MODIS and the European Space Agency's (ESA) Medium Resolution Imaging Spectrometer (MERIS) can be used to generate results similar to SeaWiFS. Another satellite that proposes similar capacities to measure Chl_{RS}-a is the Visible Infrared Imaging Radiometer Suite (VIIRS), which will be one of five instruments to fly on the National Polar-orbiting Operational Environmental Satellite System (NPOESS) Preparatory Project (NPP) spacecraft. NPP is scheduled for launch in Fall 2011. VIIRS is the successor to MODIS and SeaWiFS for Earth science data product generation. There are multiple missions planned by space agencies and overlapping of sensors that will have continuity of measurements for long-term assessment using satellite data.

5 Numeric Nutrient Criteria Development in South Florida Marine and Inland Flowing Waters

EPA is considering several methodological approaches to derive numeric criteria for nitrogen/phosphorus pollution in South Florida marine and inland flowing waters. For the purpose of this document, South Florida inland flowing waters have been defined as free-flowing, predominantly fresh surface water in a defined channel, and include, streams, rivers, creeks, branches, canals, freshwater sloughs, and other similar water bodies located in the South Florida nutrient watershed region. South Florida marine waters include estuarine and coastal waters extending three nautical miles offshore. Collectively, these waters have been classified by the State of Florida as Class I, II, or III waters as defined in Chapter 2. By and large, most of the inland flowing waters in South Florida considered by this rule making are Class III with relatively few Class I waters, while Class II and III waters make up South Florida's marine waters. The same narrative nutrient criterion applies to all of these waters: "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 and fauna." EPA is deriving numeric criteria to interpret this narrative statement to protect these waters from nitrogen/phosphorus pollution. EPA is not deriving criteria that would apply to waters located on the Seminole Indian Reservation or the Miccosukee Indian Reservation; waters located in stormwater treatment areas (STAs), wetlands, or marshes; and Class IV canals.

EPA is considering a reference condition approach to derive numeric TN and TP criteria for South Florida inland flowing waters and numeric chlorophyll *a*, TN and TP criteria in South Florida marine waters using least-disturbed sites that support balanced natural populations of aquatic flora and fauna. Alternative methods of criteria derivation for inland flowing waters that EPA is considering include stressor-response relationships between chlorophyll *a* and TN and TP, and a distributional approach using all sites. EPA is not establishing new TP criteria for canals in the Everglades Protection Area (EvPA) in deference to the Everglades Forever Act (EFA) as discussed more in Section 5.2.1. EPA is also considering derivation of TN and TP downstream protective values (DPVs) that are protective of South Florida's marine waters at the terminal reach of each tributary to the estuary or coast (see Chapter 6).

5.1 Analysis Plan for South Florida

EPA is deriving numeric criteria for nitrogen/phosphorus pollution in South Florida marine and inland flowing waters. The approach that EPA is considering is displayed in Figure 5-1.

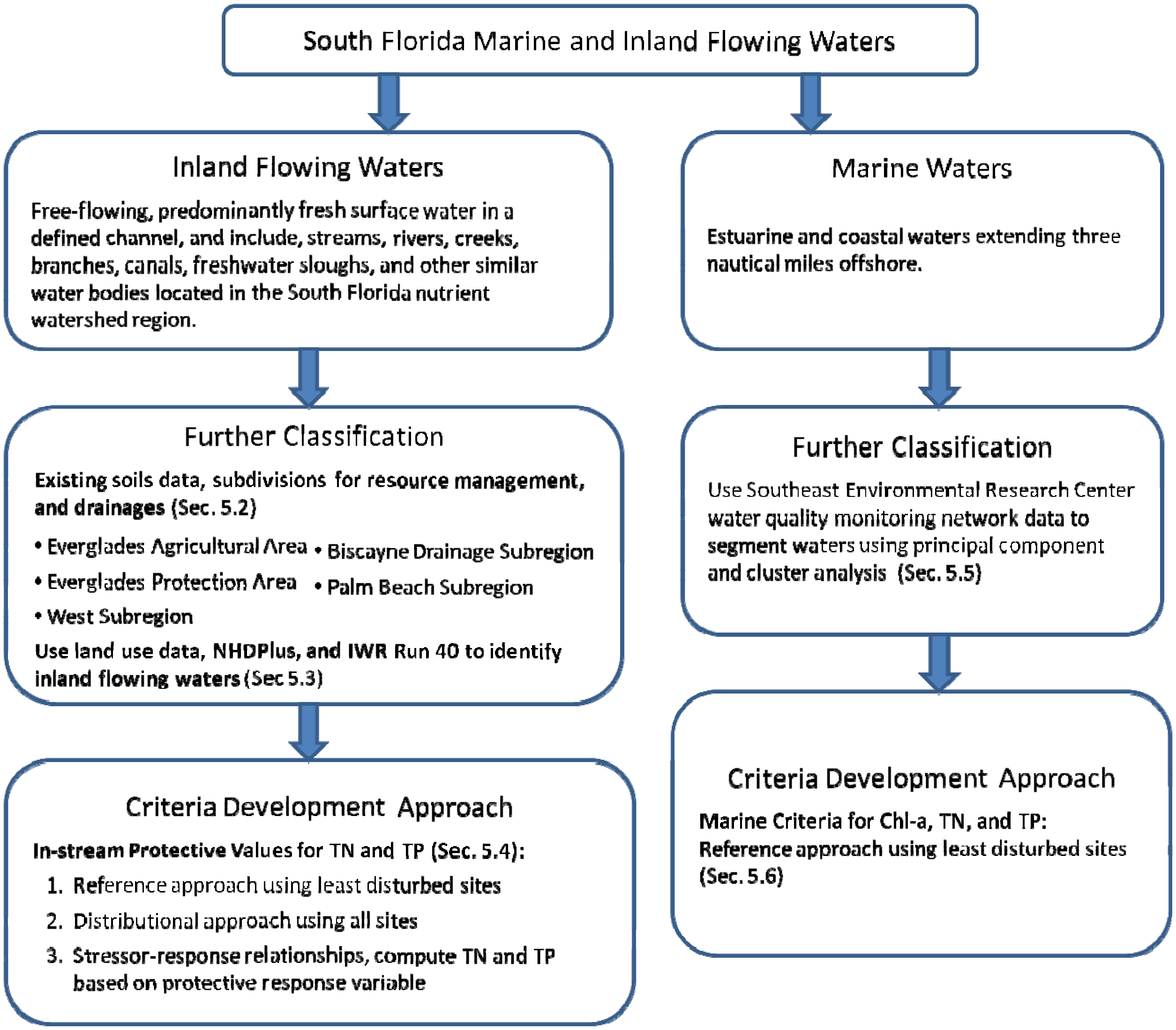

Figure 5-1. Flow chart for the development of numeric criteria for South Florida marine and inland flowing waters

5.1.1 Inland Flowing Waters

Section 5.2 describes the first step of the analysis that EPA is considering to further classify the inland flowing waters into five subregions. EPA is considering dividing South Florida into subregions to account for variability and diversity of these flowing waters. Next, EPA is considering use of the National Hydrography Dataset (NHDPlus), land use, and IWR data to develop an inventory of inland flowing waters that are subject to this rulemaking to assist in the derivation of numeric criteria (Section 5.3). This is necessary to facilitate subsequent analyses that should exclude inland waters not covered by this rulemaking such as Class IV canals.

Finally for inland flowing waters, EPA is considering derivation of instream protective values (IPVs) for TN and TP with the reference condition approach using least-disturbed sites (Section 5.4). EPA is considering identifying least-disturbed sites by integrating land use data in an approach similar to the stream corridor Landscape Development Intensity Index (LDI) developed by FDEP. In addition to the above analysis, two other analyses are being considered for developing IPVs: (1) distributional approach of all sites, and (2) stressor-response approach (Sections 5.4.1.1.3 and 5.4.1.2, respectively). Unlike the analysis which targets least-disturbed sites, the distributional approach of all sites would rely on data from all sites. The stressor-response approach would evaluate the relationships between response variables (e.g., chlorophyll *a*) and TN or TP.

5.1.2 Marine Waters

In South Florida, with its highly managed inland flows and open-water dominated systems (i.e., Florida Bay and Keys), EPA found that existing information about watershed drainages can be supplemented with data from the Southeast Environmental Research Center (SERC) water quality monitoring network to further classify marine waters. EPA considers a multivariate statistical approach to identify 30 individual waterbodies throughout the State (Section 5.5).

EPA is considering two methods for using a reference condition approach for deriving numeric criteria (Section 5.6). The reference condition approach identifies concentrations that are inherently protective of the waterbody because those concentrations are associated with currently demonstrated healthy balanced populations of aquatic flora and fauna. One approach is to estimate criteria based on an inclusive distribution of nitrogen and phosphorus values that are temporally and/or spatially averaged and are obtained from least-disturbed reference sites from a prescribed region. In this situation, EPA (2001) recommends the selection of a percentile of this distribution based on the confidence in identifying least-disturbed reference sites.

EPA is also considering an approach based on the statistical distribution of raw data and evaluated using a binomial test. In this approach, EPA would derive two criteria: (1) an average (median) concentration and (2) an upper percentile concentration. By considering an upper percentile, this second approach would be sensitive to detecting changes in the distribution of higher concentrations that may be "averaged-out" in annual geometric means.

5.2 Inland Flowing Waters: Classification

In determining the geographic scope for South Florida inland flowing waters, EPA initially focused on canals located in the Everglades ecoregion (see areas south of blue line in Figure 5-2). Because soil or substrate type at the bottom of a canal can influence the nutrient cycling and relationships between the observed biological response and nutrient levels in canals, EPA used data on soil types in South Florida, along with existing subdivisions (primarily for the Everglades Agricultural Area (EAA) and EvPA) already used for managing area resources to develop subregions. The East subregion was composed of the Southeast Coast—Biscayne monitoring basin and most of the Lake

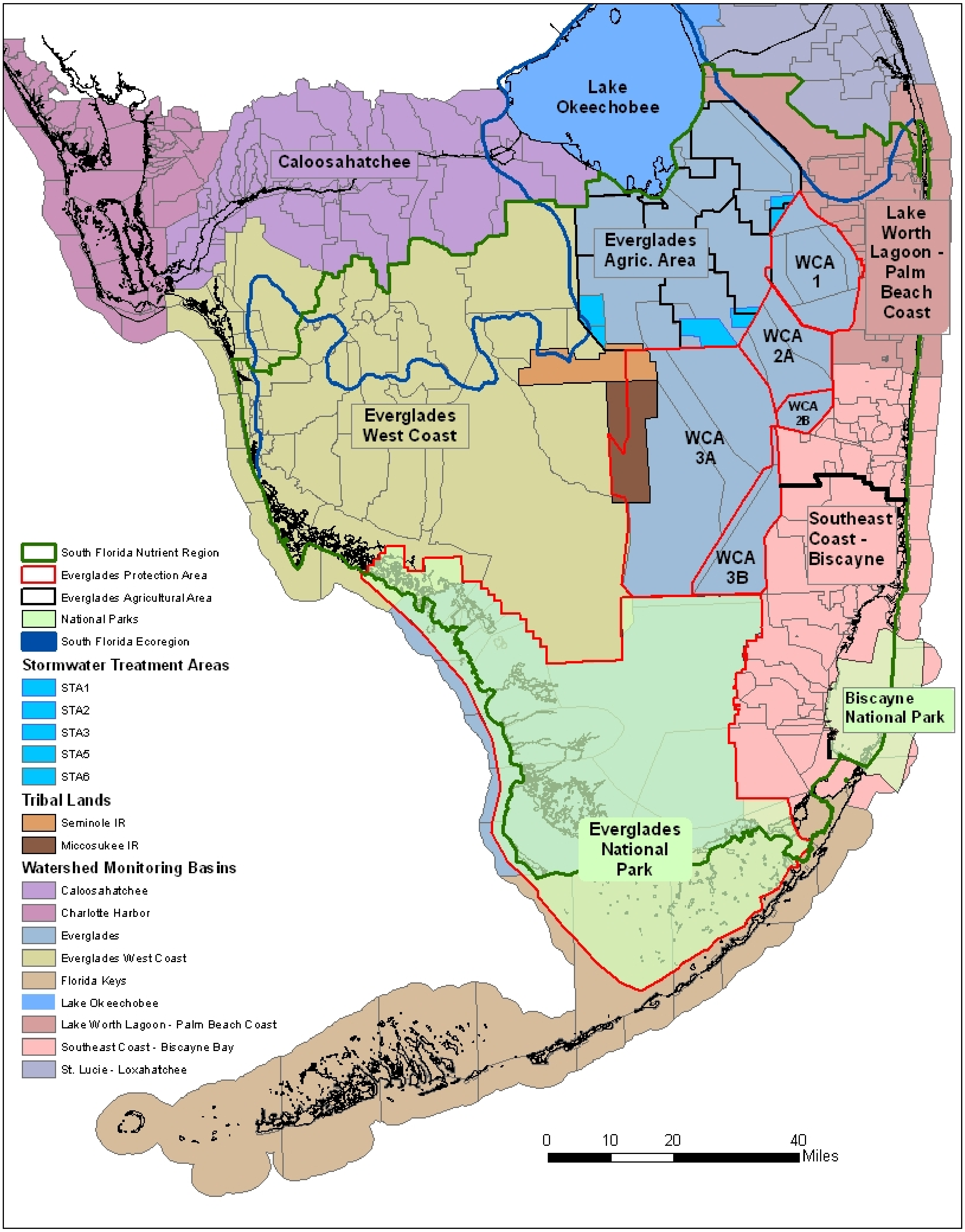

Figure 5-2. South Florida Nutrient Watershed Region

Worth Lagoon—Palm Beach Coast monitoring basin; and generally included spodosol, entisol, and alfisol soils (see Figure 5-3). The East subregion was bounded on the west by the EvPA which includes the Everglades National Park as well as additional wetlands to the north. The EvPA is largely composed of entisol soils in the south and histosol soils in the north. The EAA is located south of Lake Okeechobee and includes the northern portion of the Everglades monitoring basin not included in the EvPA. Histosol soils make up much of the EAA. The West subregion included much of Collier County and Big Cypress National Preserve, where spodosol and alfisol soils predominate.

Along the border between the Peninsula[11] and South Florida nutrient watershed regions, EPA performed spatial analyses to determine the boundary to be consistent with the monitoring basins, with exception of the area near Estero Bay on the west coast as depicted by the green line shown in Figure 5-2. EPA is also considering sub-classifying the Biscayne drainage area and analyzing it separately from the rest of the East subregion. Specifically, EPA is considering dividing the East subregion just above Canal C-9 as shown in Figure 5-4, thus allowing EPA to analyze all canals draining to the Biscayne Bay waters as one group where subregion-specific analyses are appropriate. Figure 5-5 presents a simplified map of the subregions that EPA is considering.

While the EPA will continue to consider other issues (e.g., geologic formations, groundwater interactions) for developing subregions, the EPA will initially consider, where subregion-specific analyses are appropriate, the following five subregions for developing numeric criteria for South Florida inland flowing waters:

- EAA: Everglades Agricultural Area
- EvPA: Everglades Protection Area
- West Subregion: Areas west of the EAA and EvPA
- Biscayne Drainage Subregion: Areas east of the EvPA and generally representing waters flowing toward Biscayne Bay
- Palm Beach Subregion: Areas east of the EAA and EvPA and north of the Biscayne Drainage Subregion

[11] EPA established numeric criteria for nitrogen/phosphorus pollution for flowing waters in the peninsula as well as the rest of northern Florida in a November 2010 rulemaking.

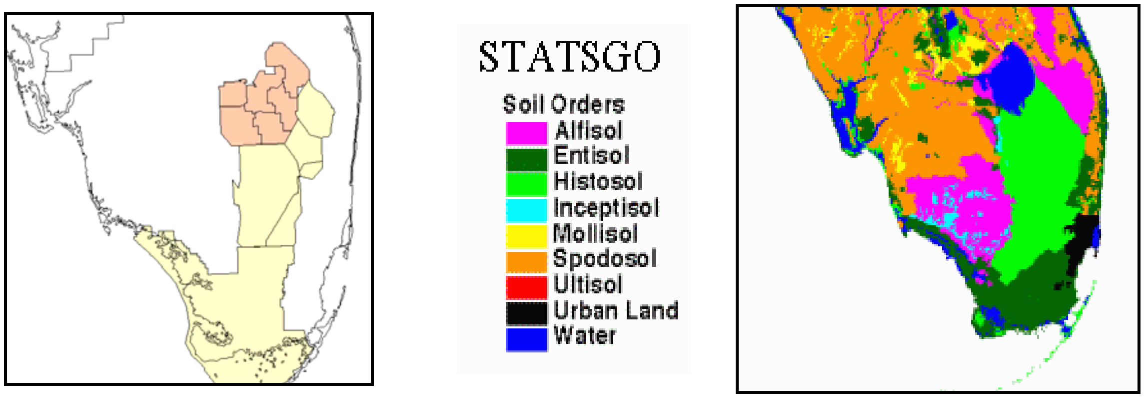

Figure 5-3. Location of EAA and EvPA relative to STATSGO soil orders for South Florida

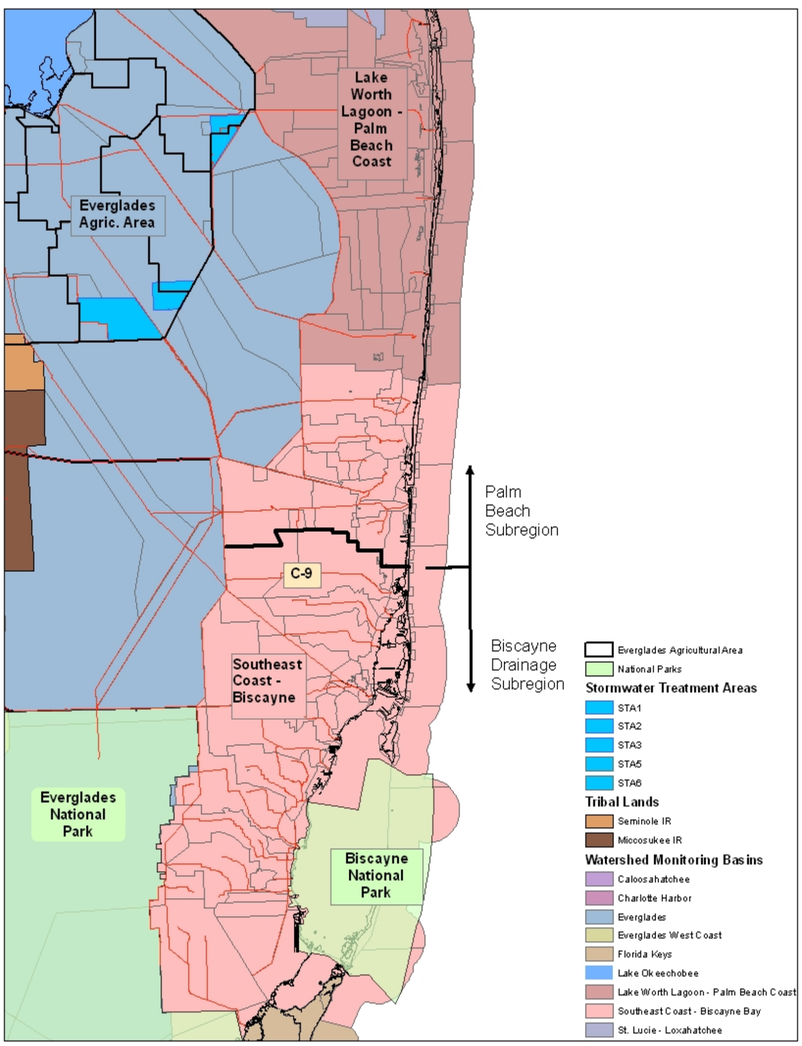

Figure 5-4. Division between Biscayne Drainage and Palm Beach subregions

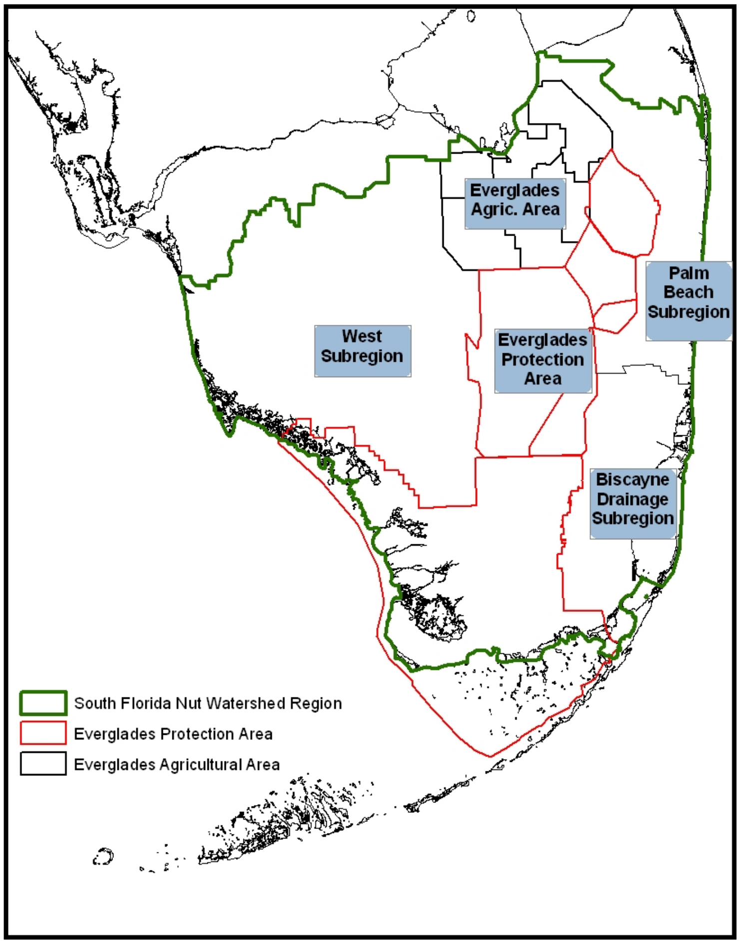

Figure 5-5. South Florida Nutrient Region subregions under consideration

5.2.1 Existing TP Criterion for the EvPA

In 1994, the EFA (Florida Statutes, Ch. 373.4592) established that "(i)n no case shall such phosphorus criterion allow waters in the Everglades Protection Area to be altered so as to cause an imbalance in the natural populations of aquatic flora or fauna." To translate this narrative, the EFA set a TP criterion of 10 parts per billion (ppb) in the EvPA. In 2005, EPA approved FDEP's implementation of the EFA TP criterion. This "numerical interpretation of the Class III narrative nutrient criterion for phosphorus" is a long-term geometric mean of TP equal to 10 ppb (or 0.010 mg/L) and applies throughout the EvPA, including Water Conservation Areas (WCAs) 2A, 2B, 3A, and 3B, as well as Everglades National Park and the Arthur R. Marshall Loxahatchee National Wildlife Refuge (WCA 1) (F.A.C. 62-302.540(1)) (see Figure 5-2). In a 2010 ruling in the case of Miccosukee Tribe of Indians of Florida vs. United States (Case 1:04-cv-21448-ASG Document 404), the U.S. District court affirmed the need to support the 10 ppb TP criterion established in the Everglades Protection Area.

EPA recognizes the aforementioned 10 ppb TP criterion as applicable to the Class III waters in the EvPA, including canals (as defined at F.A.C. 62-302.400 and F.A.C. 62-303.200(21)), and is not establishing additional numeric TP criteria for this subregion.

5.3 Inland Flowing Waters: Inventory

As mentioned above, inland flowing waters in South Florida include free-flowing, predominantly fresh surface water in a defined channel, and include streams, rivers, creeks, branches, canals, freshwater sloughs, and other similar water bodies. Pursuant to Rule 62–302.200, F.A.C., "*predominantly fresh waters shall mean surface waters in which the chloride concentration at the surface is less than 1,500 milligrams per liter.*" Also, EPA is not deriving criteria that would apply to waters located on the Seminole Indian Reservation or the Miccosukee Indian Reservation (see Figure 5-2); waters located in stormwater treatment areas (STAs), wetlands, or marshes; and Class IV canals.[12]

Currently, there does not appear to be an inventory of South Florida inland flowing waters that clearly delineates the areas described above. From the Florida Coastal Everglades Long Term Ecological Research (FCE LTER) network web site, a GIS coverage of major canals is available. Yet there are numerous smaller canals in urbanized areas that may fit the definition of Class I or II designated inland flowing waters, but are not in this coverage. NHDPlus is available from the U.S. Geological Survey. The NHDPlus provides extensive GIS data on canals and other surface water systems; however, these data do not distinguish Class IV canals from other waters. FDEP provided EPA a coverage of canals identified as Class IV canals. While initial review of the Class IV canals suggests that this coverage matches well with NHDPlus, it appears limited to portions of the EAA and areas to the west of the EAA and thus may not be comprehensive. EPA also has access to the detailed land use data maintained by FDEP and individual Water Management Districts.

EPA is considering integrating these data to develop an inventory of inland flowing waters for the derivation of numeric criteria. The purpose of this inventory is to serve as a tool for screening water quality monitoring and other data from South Florida flowing waters that are designated as Class I and III from data collected from those waters that are designated as Class IV (Agricultural Water Supplies). As mentioned above, EPA is not establishing numeric criteria for flowing waters designated as Class IV. An inventory of canals will also provide technical and management staff with an improved understanding of the complexity associated with canal management. Distinguishing natural streams from canals might allow EPA to assess whether the numeric criteria for canals should be set differently from natural streams.

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[12] The definition of Class IV is provided in Rule 62-302.400: All secondary and tertiary canals wholly within agricultural areas are classified as Class IV and are not individually listed as exceptions to Class III. "Secondary and tertiary canals" shall mean any wholly artificial canal or ditch which is behind a control structure and which is part of a water control system that is connected to the works (set forth in Section 373.086, F.S.) of a water management district created under Section 373.069, F.S., and that is permitted by such water management district pursuant to Section 373.103, 373.413, or 373.416, F.S. Agricultural areas shall generally include lands actively used solely for the production of food and fiber which are zoned for agricultural use where county zoning is in effect. Agricultural areas exclude lands which are platted and subdivided or in a transition phase to residential use.

To develop the inventory, NHDPlus flow line coverage (see Figure 5-6) would be intersected with the land use data. In general terms, NHDPlus flow line segments with adjoining non-agricultural land uses would be classified as Class III. Canals that coincide with the LTER coverage of major canals would be designated as major canals. Segments classified as canal/ditch, artificial path, or connector with only adjoining agricultural land uses would be classified as potential Class IV canals. EPA considers this second class as *potential* Class IV since there are additional requirements that must be met to qualify as a Class IV canal such as being behind a control structure. Water quality data from these potential Class IV segments would not be included in the derivation of the numeric criteria. The process of identifying Class IV segments using the above approach would be compared to the independently provided Class IV canal data coverage. Note that in Figure 5-6, the pink lines shown south of Lake Okeechobee are from the GIS coverage of Class IV canals provided by FDEP. Most of these "pink lines" coincide with NHDPlus flowlines identified as canal/ditch. This comparison is more clearly portrayed in Figure 5-7, which is a smaller scale map that focuses on the area near STAs 5 and 6 and the Seminole Indian Reservation. In the upper portion of Figure 5-7, the (pink) Class IV canals coincide (i.e., lie on top of) with the (green) NHDPlus canals. EPA's approach for identifying potential Class IV canals from NHDPlus would include the canals where the (pink) Class IV canals coincide with the NHDPlus canals.

The NHDPlus flow line coverage would also be used to identify natural streams. Based on visual inspection of the NHDPlus flow line coverage, most of the segments classified as streams are near the coastline, which indicates that they might not be predominately fresh water.

As mentioned before, predominantly fresh waters have been defined in this effort as having chloride concentrations less than 1,500 mg/L chloride at the surface, pursuant to Rule 62–302.200, F.A.C. Thus, waters greater than 1,500 mg/L chloride at the surface will be classified as estuarine or marine waters and subject to analyses considered elsewhere in this methods document. Based on a preliminary review of data from FDEP's IWR, chloride data are sparse in South Florida. Thus, EPA is also considering the use of the relationship between specific conductivity and salinity to identify predominately fresh water. Pending these analyses, natural streams such as Cocohatchee River, Gordon River, Turner River, and Henderson Creek near Naples, Florida and Halfway Creek in the Ten Thousand Island area might be included in the analyses considered in this chapter if they are predominantly fresh water. If they are determined to be predominantly marine, such waters will analyzed as part of the estuarine criteria derivation effort described in Chapter 3. For canals, additional information such as control structure locations may be used to confirm these analyses. Additional consideration may be necessary to identify sloughs that are in defined channels.

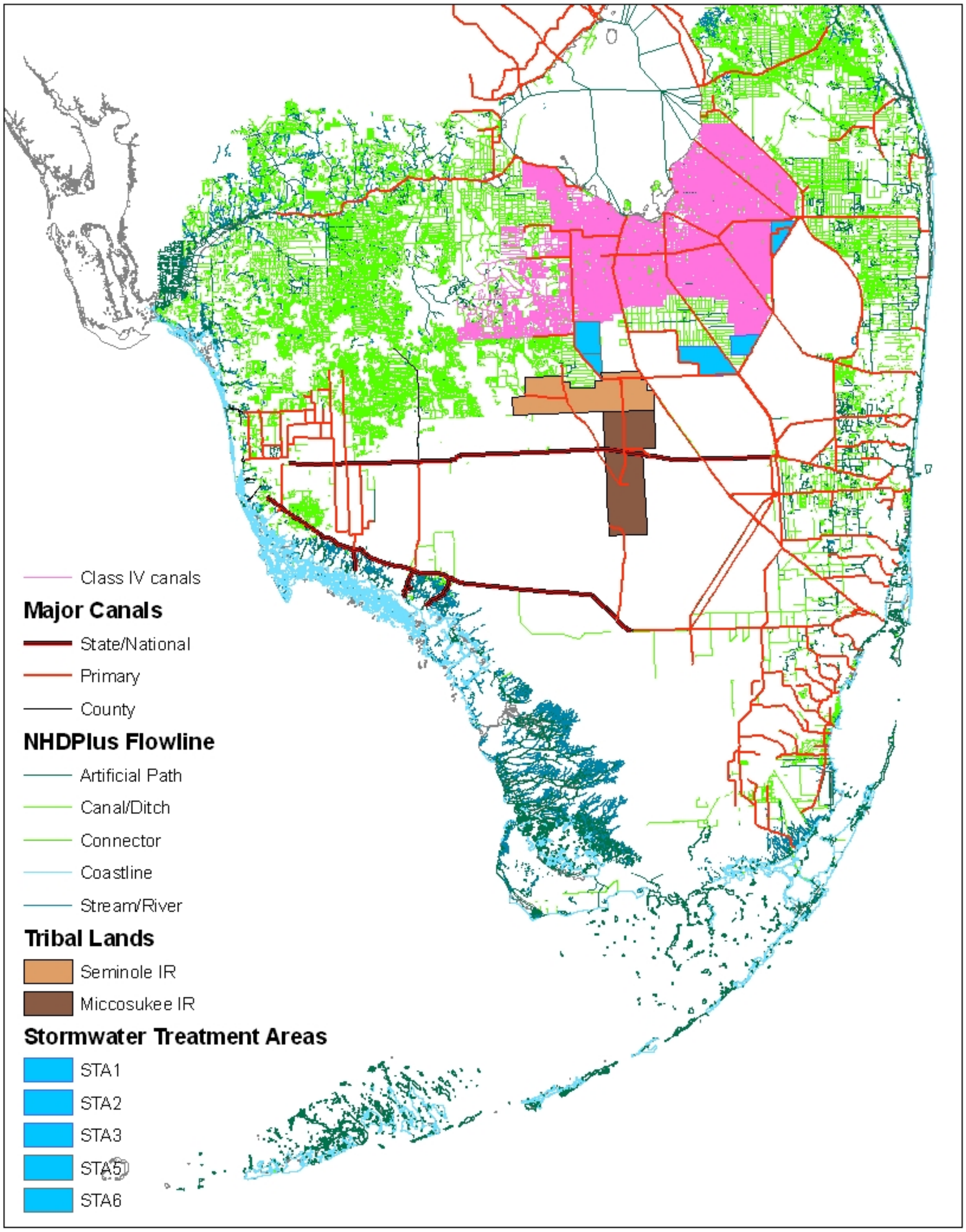

Figure 5-6. NHD flowline data in South Florida. Note that the pink lines shown south of Lake Okeechobee are from the GIS coverage of Class IV canals provided by FDEP. Most of these "pink lines" coincide with NHDPlus flowlines identified as canal/ditch.

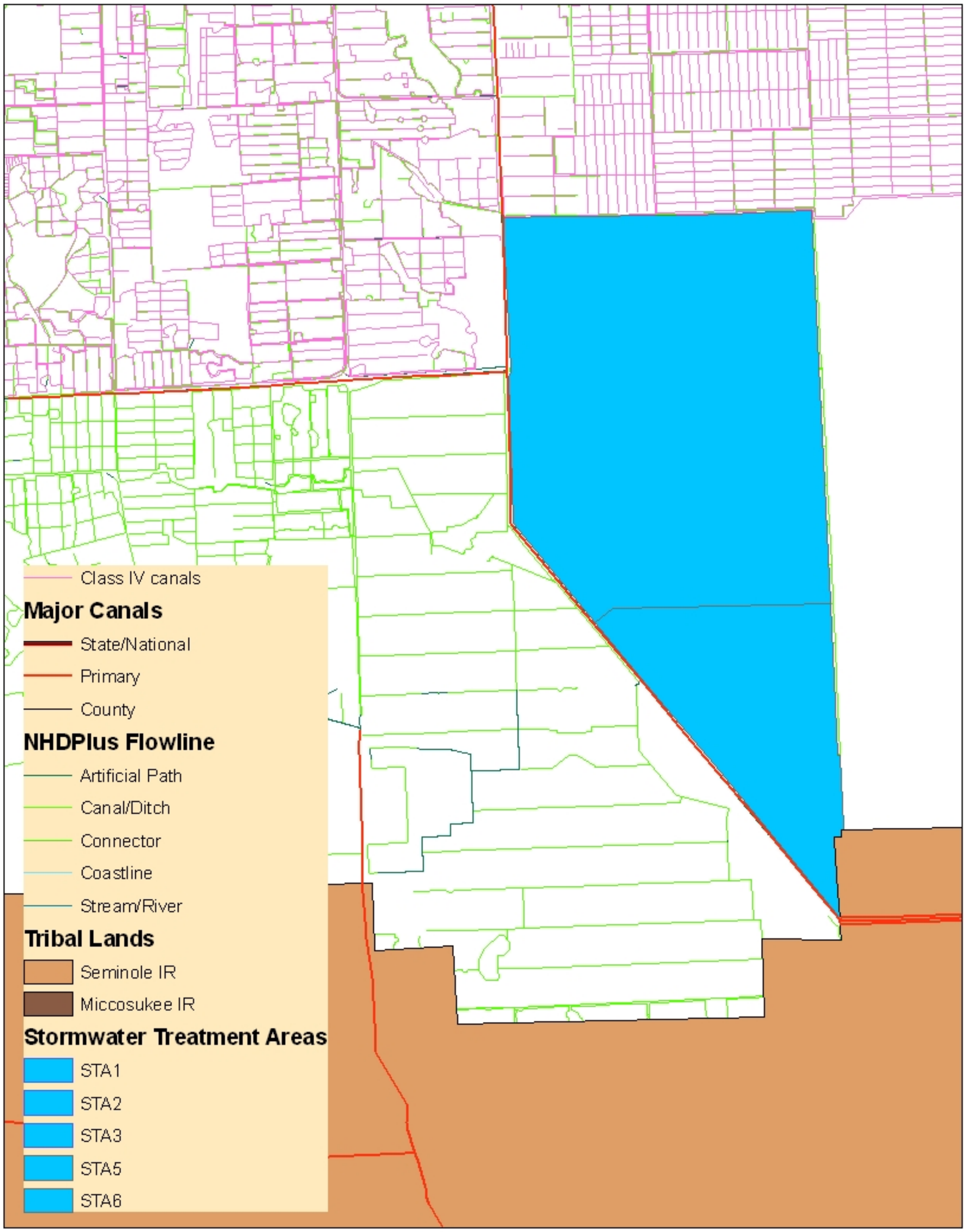

Figure 5-7. NHD flowline data near Stormwater Treatment Areas 5 and 6 and Seminole Indian Reservation

5.4 Inland Flowing Waters: Data Sources and Assessment Endpoints

EPA has considered a range of approaches for deriving numeric criteria for inland flowing waters in South Florida. One approach EPA is considering for developing instream protective values (IPVs) for TN and TP is the reference condition approach using least-disturbed sites (Section 5.4.1.1). The applicable assessment endpoint for this approach is balanced faunal communities as described in Chapter 2.

Specifically for these waters, EPA is considering the use land use data of a multi-metric index of aquatic macroinvertebrates (stream condition index or SCI) modified for use in these highly managed systems as a means to indicate balance in the natural populations of aquatic flora and fauna. Snyder et al. (1998), described a modified version of FDEP's SCI bioassessment methodology[13], adapted to characterize and monitor benthic macroinvertebrates in Dade County freshwater canal systems. Dade County's monitoring program involved 36 canal sites, and focused on the sampling of potential reference site locations as well as "areas of known impairment." Dade County canal bioassessment results, segregated by land use designations, indicated a gradient of biological condition reflected in SCI scores ranging from poorest in urban/industrial, suburban, and agricultural canals, to good in canals surrounded by wetlands (Figure 5-8).

Based on the findings in Snyder et al. (1998), EPA is considering a land use-based reference condition approach for deriving numeric criteria for canals and other inland flowing waters in South Florida. EPA is considering using quantitative land use analyses to identify least-disturbed sites (e.g., sites surrounded by wetlands), which in turn can serve as a reference condition population for deriving numeric criteria (see Section 5.4.1.1).

[13] Metrics included in Dade County's canal SCI were a subset of seven of the ten that make up FDEP's SCI. All metrics except the percent filterers and EPT taxa metrics, which Dade County found to be "very low" at all of their sampled sites. With a maximum possible score of 5 assigned to each metric, Dade County's canal SCI spans a range of 0 to 35, in contrast with FDEP's SCI which spans a range of 0 to 100.

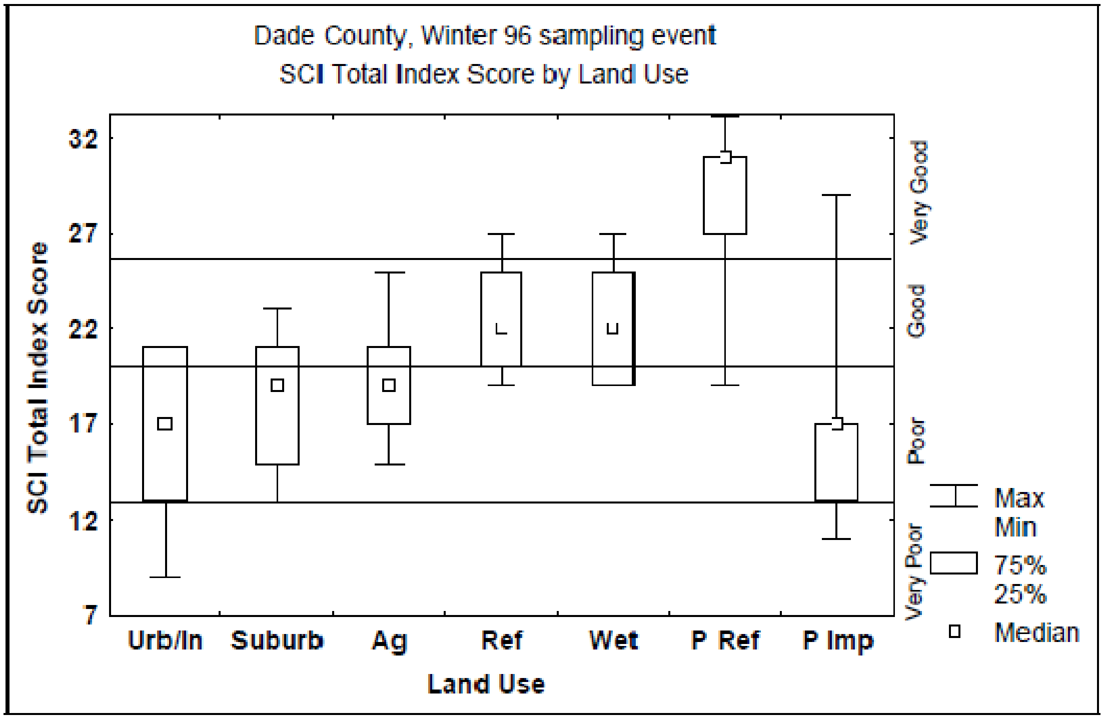

Figure 3-18. Dade County canal bioassessment results (winter 1996) segregated by land use designation. Land uses: urban/industrial (urb/in); suburban (suburb); agricultural (Ag); canals surrounded by wetlands (wet). Note: Ref = wetland canals with highest habitat index values; P Ref = DEP reference conditions for peninsular Florida; P Imp = DEP impaired site results for peninsular Florida.

Figure 5-8. Box plots of SCI total index scores for a variety of land uses (source: Snyder et al. 1998)

The second approach that EPA is considering is a stressor-response approach (Section 5.4.1.2). In this approach, empirical relationships between response variables (e.g., chlorophyll *a*) and nutrients would be developed. EPA is also considering means to determine a protective chlorophyll *a* level for canal systems as EPA has found chlorophyll *a* to be an appropriate response variable for flowing waters and lakes to measure the assessment endpoint of balanced phytoplankton biomass and production. Yet, this may prove difficult since there is limited evidence to assist in the identification of a specific protective chlorophyll *a* threshold in canals.

The approach for identifying least-disturbed sites integrates land use data into its assessment. EPA is also considering using data from the following sources:

- IWR Run 40
- South Florida Water Management District (i.e., DBHYDRO)
- Local agencies with water quality monitoring data
  - Broward County Department of Planning and Environmental Protection
  - Miami-Dade Department of Environmental Resources Management

EPA is considering applying both approaches using the inventory of inland flowing waters and subregions described earlier in this chapter. Based on IWR Run 40 database, there are more than 50,000, 65,000, and 9,500 observations of TN, TP, and Chl-a from 1990 to 2010 for the South Florida nutrient watershed region, respectively. The total number of observations actually available for analysis of inland flowing waters may be more or less depending on the addition of data from other data sources not already in IWR and removing data from the analysis,such as observations from stations located on Class IV canals.

5.4.1 Deriving Numeric Criteria

5.4.1.1 Reference Condition Approach—Least-Disturbed Sites

As mentioned above, one approach that EPA is considering is the reference condition approach (USEPA 2000c). A reference condition for nutrients can be determined using a variety of empirical approaches. They include correlative, stressor-response relationships between a biological response variable and TN or TP, or a statistical distribution of nutrient data. The latter reference condition approach involves setting criteria based on an inclusive distribution of values (e.g., temporally and/or spatially averaged) obtained from least-disturbed reference sites from a prescribed region (based on climate and geology). In this case, EPA is considering using an upper percentile value of this distribution to represent a level of nitrogen and phosphorus concentrations that will inherently support aquatic life. This methodology will ensure that nutrient concentrations associated with biologically healthy, well-balanced communities would be considered acceptable as protective of those communities. The percentile chosen will be dependent upon the level of confidence that EPA has that these sites are in fact, the least-disturbed conditions. The reference condition approach is an inference model that is used to derive numeric criteria based on conditions that are known to result in a balance of the natural populations of aquatic flora and fauna in a particular set of waterbodies. Developing criteria from a concentration distribution of least-disturbed reference condition sites, results in criteria that are inherently protective because those concentrations are associated with demonstrated healthy biological communities.

Since the type of soil or substrate can influence the relationships between observed biological response and nutrient levels in canals, EPA is considering whether to perform these analyses at a subregion level as well as for the entire nutrient region. Also, EPA will continue to investigate other hydrogeological issues that may merit further refinement of the subregions.

5.4.1.1.1 Selecting Least-Disturbed Sites

EPA is considering using the LDI to identify a population of sites that are least-disturbed. The LDI may be used in conjunction with or separately from other screening criteria, such as applicable CWA section 303(d) impairments to identify least-disturbed sites.

The LDI is a quantitative method to evaluate the intensity of human usage of landscapes, based on nonrenewable energy flow. The application of this method is based on the principle that the "intensity of human-dominated land uses in a landscape affects ecological processes of natural communities," where more intense activities result in greater effects on ecological processes (Brown and Vivas 2005). The LDI is calculated as the area-weighted value of land uses within an "area of influence," where each land use is assigned a land use coefficient derived by Brown and Vivas (2005) (Table 5-1), multiplied by the percent total area occupied by each land use (determined by GIS land use coverage developed from high-resolution aerial photographs), and summed using Equation 5-1 (Brown and Vivas 2005)

$$LDI_{total} = \sum \%LU_i * LDI_i \qquad \text{Equation 5-1}$$

where

LDI_{total} = LDI ranking for landscape unit

$\%LU_i$ = percent of the total area of influence in land use i

LDI_i = landscape development intensity coefficient for land use i (see Table 5-1 for coefficients).

Table 5-1. Land use classifications and LDI coefficients. Higher values indicate greater intensity of human land use (adapted from Brown and Vivas 2005).

Land use	LDI coefficients
Natural system	1.00
Natural open water	1.00
Pine plantation	1.58
Recreational / open space – low-intensity	1.83
Woodland pasture (with livestock)	2.02
Improved pasture (without livestock)	2.77
Improved pasture – low-intensity (with livestock)	3.41
Citrus	3.68
Improved pasture – high-intensity (with livestock)	3.74
Row crops	4.54
Single family residential – low-density	6.9
Recreational / open space – high-intensity	6.92
Agriculture – high intensity	7.00
Single family residential – medium density	7.47
Single family residential – high density	7.55
Mobile home (medium density)	7.70
Highway (- lane)	7.81
Low-intensity commercial	8.00
Institutional	8.07
Highway (4-lane)	8.28
Mobile home (high density)	8.29
Industrial	8.32
Multi-family residential (low rise)	8.66
High-intensity commercial	9.18
Multi-family residential (high rise)	9.19
Central business district (average 2 stories)	9.42
Central business district (average 4 stories)	10.00

As discussed in FDEP's Nutrient Plan (2009a), 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 (see example Figure 5-9) 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 2009b; Fore 2004; Fore et al. 2007; Niu 2004). Furthermore, it has been demonstrated that an LDI of 2.0 is an appropriate and biologically significant break point that can be used to distinguish benchmark reference conditions from potentially disturbed areas.

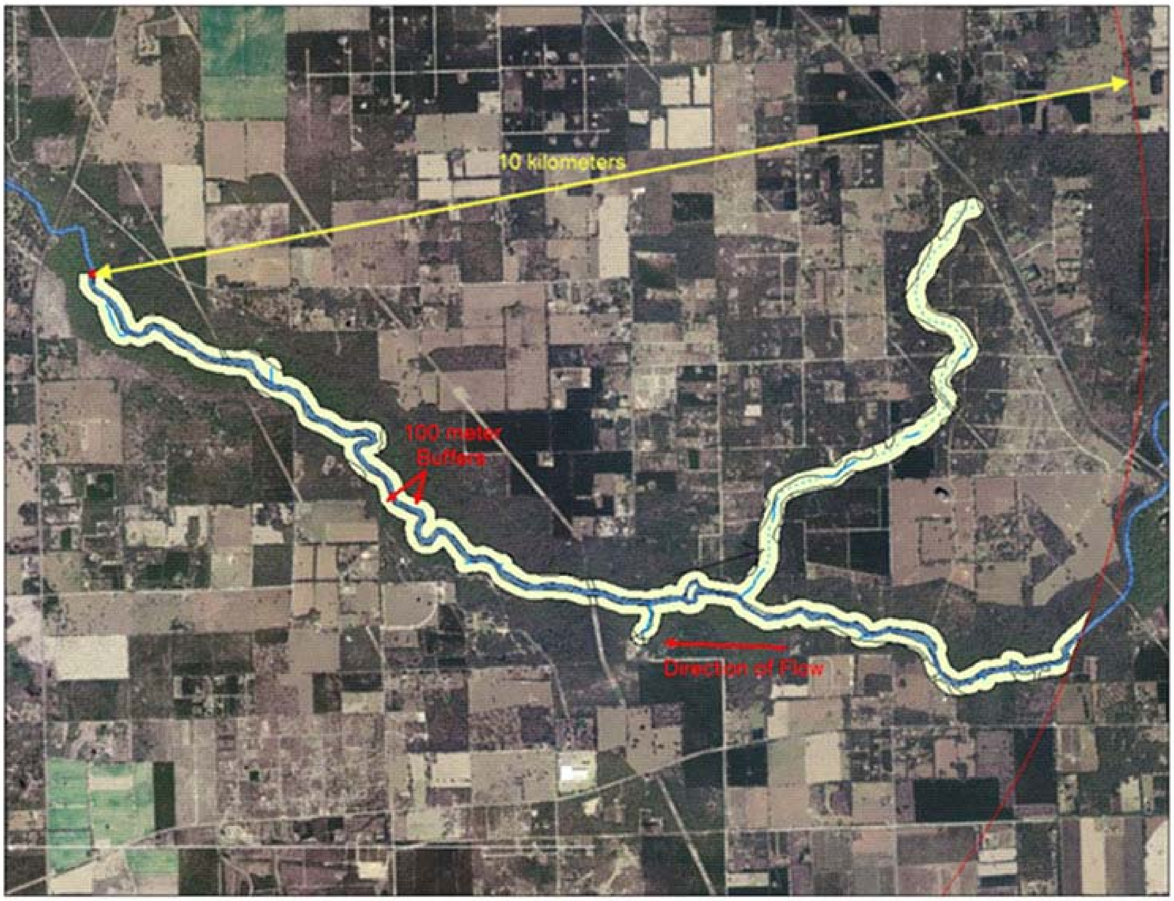

Figure 5-9. Depiction of land use area (light yellow) included in an LDI calculation

EPA is considering the use of a methodology similar to FDEP's stream corridor LDI when selecting least-disturbed sites in the South Florida nutrient watershed region. Using the land use data maintained by FDEP or at individual water management districts, the land use within a 100 m buffer of each canal segment over the length of the canal segment would be tabulated and converted into a canal LDI. This LDI would be compared to in-canal nitrogen and phosphorus concentrations to determine if the LDI can be used to explain a significant portion of the variability. Depending on preliminary results, EPA may need to adjust the size of the canal buffer and methods for considering the influence of land beyond the immediate segment.

5.4.1.1.2 Annual Average Concentration

Aquatic life water quality criteria include three components: magnitude, frequency, and duration. One approach is to consider the annual geometric mean concentration for the magnitude component of water quality criteria (implicitly using an annual duration). The geometric mean is preferred to the arithmetic mean because the geometric mean is a better estimator of the distribution central tendency when the data are log-normally distributed which is common for parameters under consideration herein. The annual geometric mean can be computed by taking the natural logarithm of the concentrations, and then compute the mean by station and year. From these values, an overall mean, $\bar{x}$, and variability associated with the factors of year and station can be estimated with standard statistical software (e.g., Minitab, SAS, R, SYSTAT).

In past efforts, the 75th or the 90th percentile of this distribution has been selected based on the level of confidence EPA has that the sites are, in fact, least-disturbed. Using the 75th percentile annual geometric mean (i.e., the criteria magnitude) as an example, EPA would compute the result as

$$GM_{75} = e^{(\bar{x} + 0.674 \cdot s)}$$

where 0.6745 is the inverse of the standard normal cumulative distribution with a probability of 0.75 and s is the corresponding standard deviation. (Substitute 1.2816 to compute GM_{90}.)

Consideration should also be given to how often the test concentration can exceed the criteria magnitude and still meet the applicable designated uses. When considering annual geometric means, most dialogue on frequency ranges from one in three years to two in five years as meeting designated uses. If there are no changes in water quality, one would expect that the GM_{75} would be exceeded, on average, 25 percent of the time. Table 5-2 presents the calculated probability that the proportion of years not meeting the criteria is greater than (A) 10 percent, (B) 20 percent, and (C) 25 percent of years not meeting the criteria. If the annual geometric mean exceeded the GM_{75} in two of three years, one would be 84 percent confident that that annual geometric mean of the test data exceeded GM_{75}. If the annual geometric mean exceeded the GM_{75} in three of five years, one would be 90 percent confident that that annual geometric mean of the test data exceeded GM_{75}. Several different formulations of the percentile of the annual geometric mean and frequency term could be considered from a statistical perspective. One potential concern with formulations that would permit the annual geometric mean to exceed the criteria magnitude in two or more consecutive years is that protracted adverse water quality conditions could cause impairment of designated uses.

Table 5-2. (A) The calculated probability that the population proportion of years not meeting the criteria is greater than 10%, based on the observed numbers of exceedances, (B) calculated probability that the population proportion of years not meeting the criteria is greater than 20%, based on the observed numbers of exceedances, (C) calculated probability that the population proportion of years not meeting the criteria is greater than 25%, based on the observed numbers of exceedances. Values are colored in a stoplight pattern indicating probabilities less than 80% (green), 80-90% (orange), and greater than 90% (red)

(A) Probability that >10% Exceed Criteria

Years	Observed Exceedances					
	0	1	2	3	4	5
2	0.00	0.81	0.99			
3	0.00	0.73	0.97	1.00		
4	0.00	0.66	0.95	1.00	1.00	
5	0.00	0.59	0.92	0.99	1.00	1.00
6	0.00	0.53	0.89	0.98	1.00	1.00

(B) Probability that >20% Exceed Criteria

Years	Observed Exceedances					
	0	**1**	**2**	**3**	**4**	**5**
2	0.00	0.64	0.96			
3	0.00	0.51	0.90	0.99		
4	0.00	0.41	0.82	0.97	1.00	
5	0.00	0.33	0.74	0.94	0.99	1.00
6	0.00	0.26	0.66	0.90	0.98	1.00

(C) Probability that >25% Exceed Criteria

Years	Observed Exceedances					
	0	1	2	3	4	5
2	0.00	0.56	0.94			
3	0.00	0.42	0.84	0.98		
4	0.00	0.32	0.74	0.95	1.00	
5	0.00	0.24	0.63	0.90	0.98	1.00
6	0.00	0.18	0.53	0.83	0.96	1.00

5.4.1.1.3 Distribution Approach—All Sites

Another approach that EPA is considering is a distributional approach of all nitrogen and phosphorus concentrations. In this application, distributions of nitrogen or phosphorus data are assembled from all sites in the subregion (or, alternatively, a random selection of sites). Assuming the pool of sites in the distribution is large enough to represent all waters in the subregion and reasonably reflect the full range of ambient conditions with a disturbance gradient from least to most impacted, the selection of a lower percentile of this distribution, should provide a nutrient level that can be assumed to inherently support a balanced natural population of aquatic flora and fauna. Several examples with data from actual reference sites have supported the use of the selection of a lower percentile of this distribution as being protective of the designated use (USEPA 2000b, 2000c). Computationally, the approach is identical to that described in Section 5.4.1.1 except that the data are not screened for least-disturbed sites.

5.4.1.2 Stressor-Response Relationships

Another approach described in previously published EPA guidance for deriving numeric criteria (USEPA 2000c) is to quantify a stressor-response relationship between nitrogen/phosphorus and adverse effects on aquatic life, and then use that relationship to establish numeric criteria. The observed dose-response relationship could be described by a model (e.g., trophic state classification, regional predictive model, nutrient sensitive biocriteria), which in turn would link nutrient concentrations to the relative risk of environmental harm. Numeric criteria would be based on a correlative relationship between nutrients and biological responses at a level that would support balanced natural populations of aquatic flora and fauna. Although the underlying relationship certainly exists, in nature it has often been difficult to quantify because of the complexity of nutrient dynamics, the influence of interfering factors, and the temporal and/or spatial scales of the data and analyses.

In this analysis, matched chlorophyll *a* and nutrient data would be analyzed using regression. Using the matched chlorophyll *a* and nutrient data (see Figure 5-10 for site locations) assembled, Figure 5-11 is a bivariate plot of annual geometric mean chlorophyll *a* as function of TP together with the fitted regression equation and 90 percent prediction interval. Figure 5-12 displays the residual plots. This regression model explains 49 percent of the variance; the residual plots suggest that the residuals have a constant variance, are independent, and are normally distributed. Overall, this model might be appropriate provided that a chlorophyll *a* criterion concentration can be selected that supports balanced natural populations of aquatic flora and fauna.

Figure 5-13 is a bivariate plot of annual geometric mean chlorophyll *a* as function of TN with the fitted regression equation along with the 90 percent prediction interval. Figure 5-14 displays the residual plots. This regression model explains 16.5 percent of the variance, and the residual plots suggest that the residuals are normally distributed. EPA acknowledges the visual appearance of a higher variance in the middle of the TN range, but further investigation of the lowest and highest chlorophyll *a* concentrations did not reveal anomalies meriting their exclusion. Because of the residual variance heteroscedasticity and low overall R^2 value, this model is not as strong as the TP model. Using this data, EPA may consider whether there is additional data available to control for the effects of stressors other than nitrogen in these waters and may instead consider relying on the reference condition approach described previously to derive TN criteria.

Alternately, it is possible to use quantile regression. In this application, quantile regression computes an estimate of the 90^{th} conditional quantile function of the response, given the stressor (TN or TP). The function presumes a linear specification for the quantile regression model, i.e., that the formula defines a model that is linear in parameters. The function minimizes a weighted sum of absolute residuals that can be formulated as a linear programming problem. To minimize the impact of skewed data, the $\log_{10}$ (logarithm, base 10) transformation can be computed before applying the quantile regression algorithm. Figure 5-15 presents bivariate plots of chlorophyll *a* as a function of TN and TP (in $\log_{10}$ space) overall. Each plot displays the Spearman's rho correlation coefficient, the locally weighted scatterplot smoothing (LOESS)[14] line (black line), and the quantile regression results (blue line and blue text for the regression terms). Should EPA chose to use stressor-response models to develop numeric criteria; EPA will use quantitative metrics for evaluation of these models.

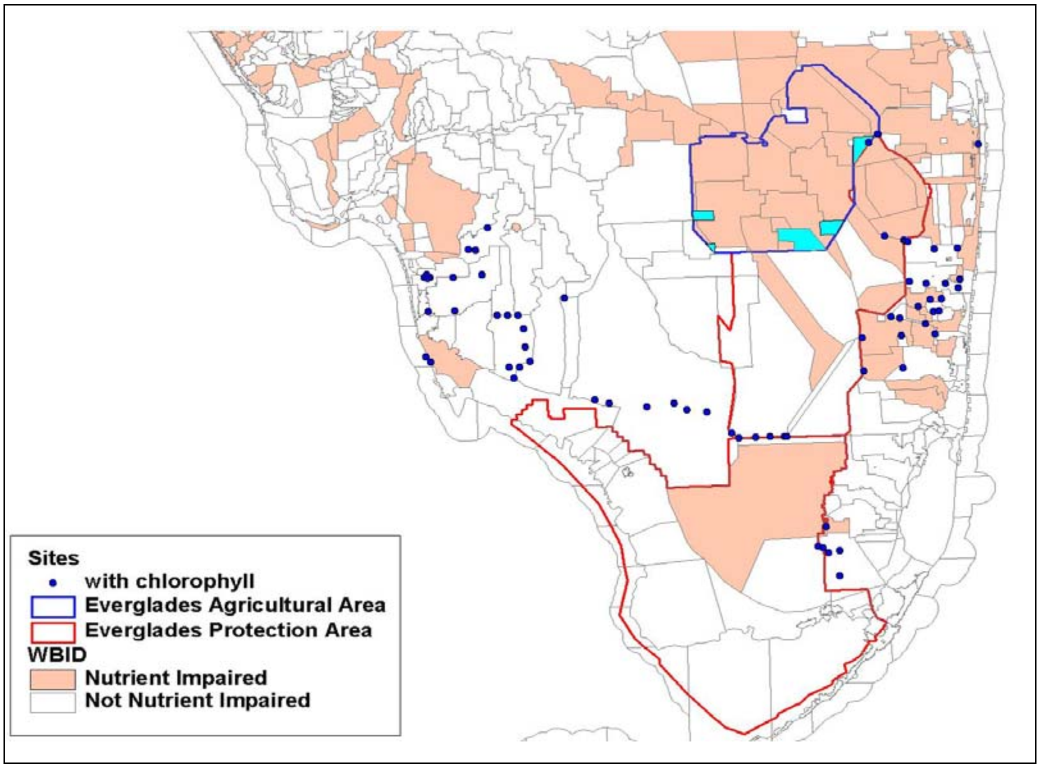

Figure 5-10. Site locations (n=72) with chlorophyll *a* data evaluated in this document together with Florida WBIDs that FDEP has identified as nutrient-impaired

[14] We used a locally weighted scatterplot smoothing (LOESS) technique to visually examine possible trends along environmental gradients. The LOESS technique (Cleveland 1979) models nonlinear relationships where linear methods do not perform well, and fits simple models to localized subsets of the data to construct a function that describes, essentially, the central tendency of data along nutrient gradients. We used a smoothing factor of 0.7 to compute LOESS curves.

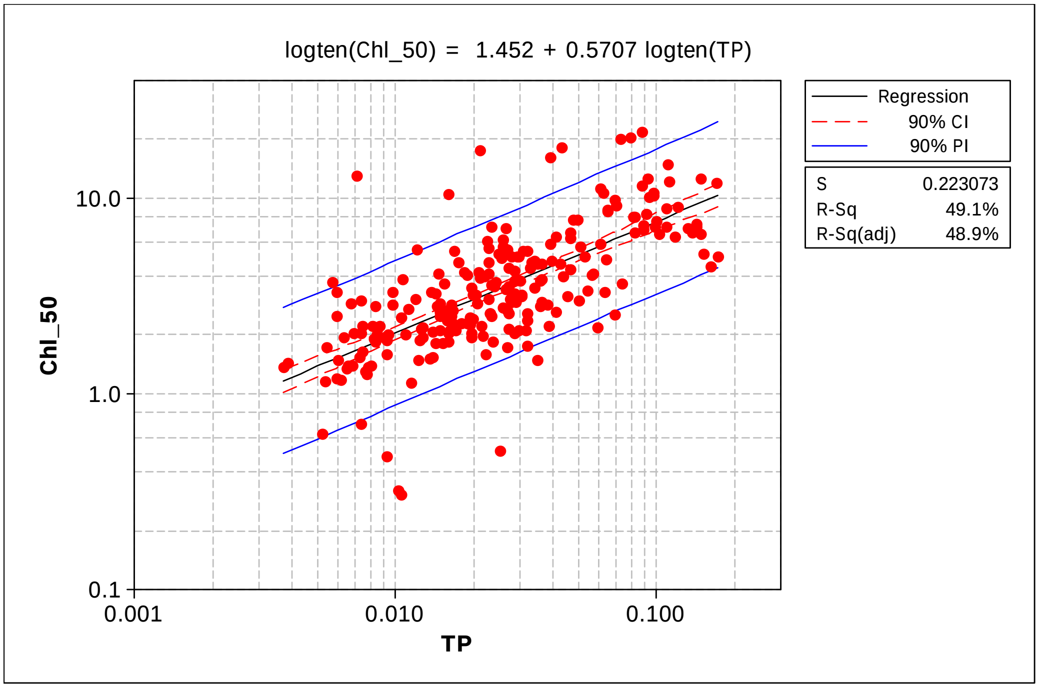

Figure 5-11. Bivariate plot of annual geometric mean chlorophyll *a* as a function of TP with fitted linear regression and 90% prediction interval

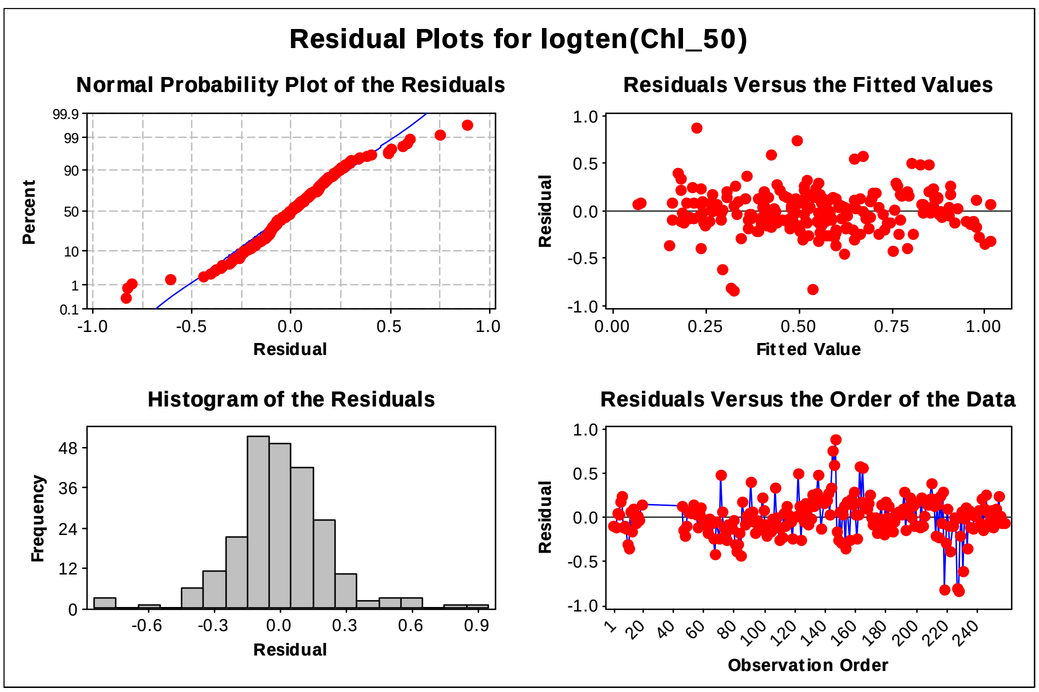

Figure 5-12. Residual plots from chlorophyll *a*/TP regression analysis

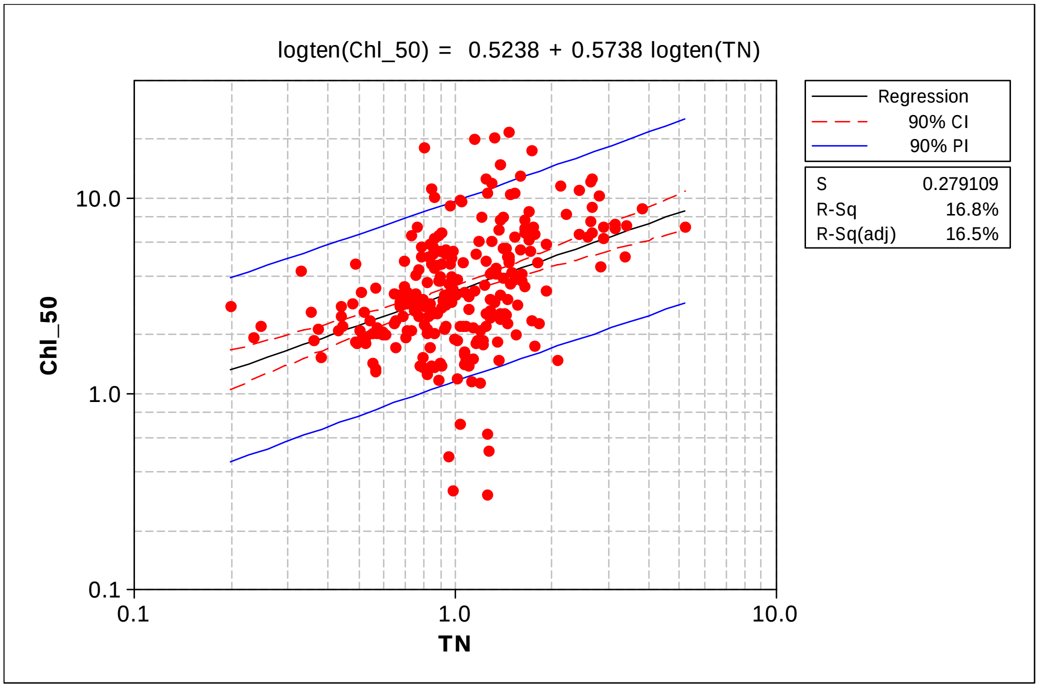

Figure 5-13. Bivariate plot of annual geometric mean chlorophyll *a* as a function of TN with fitted linear regression and 90% prediction interval

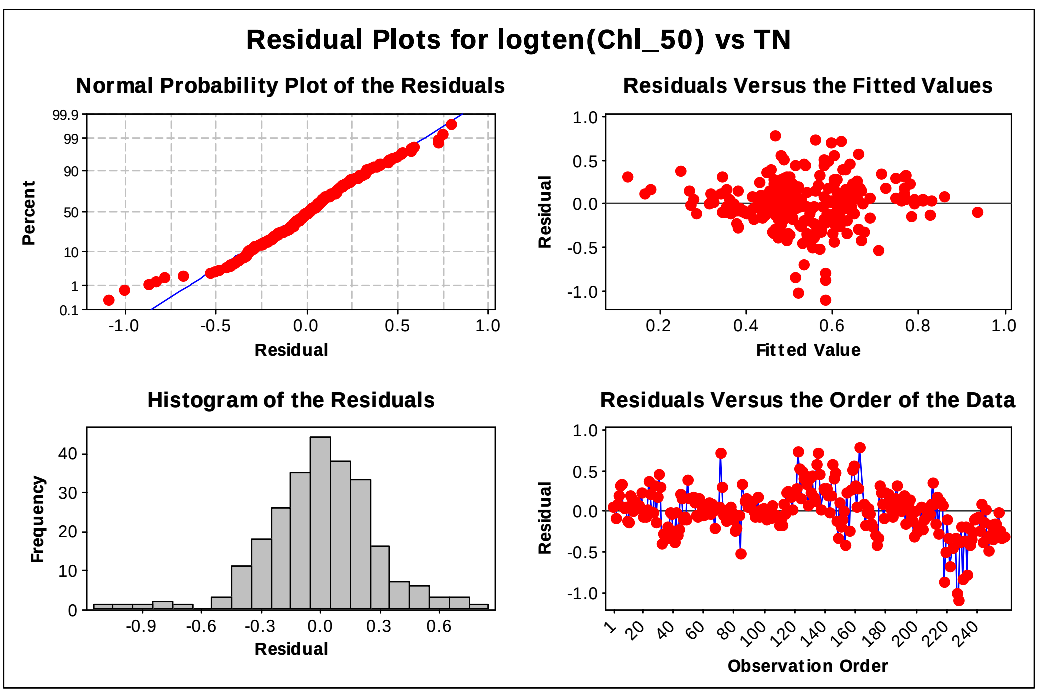

Figure 5-14. Residual plots from chlorophyll *a*/TN regression analysis

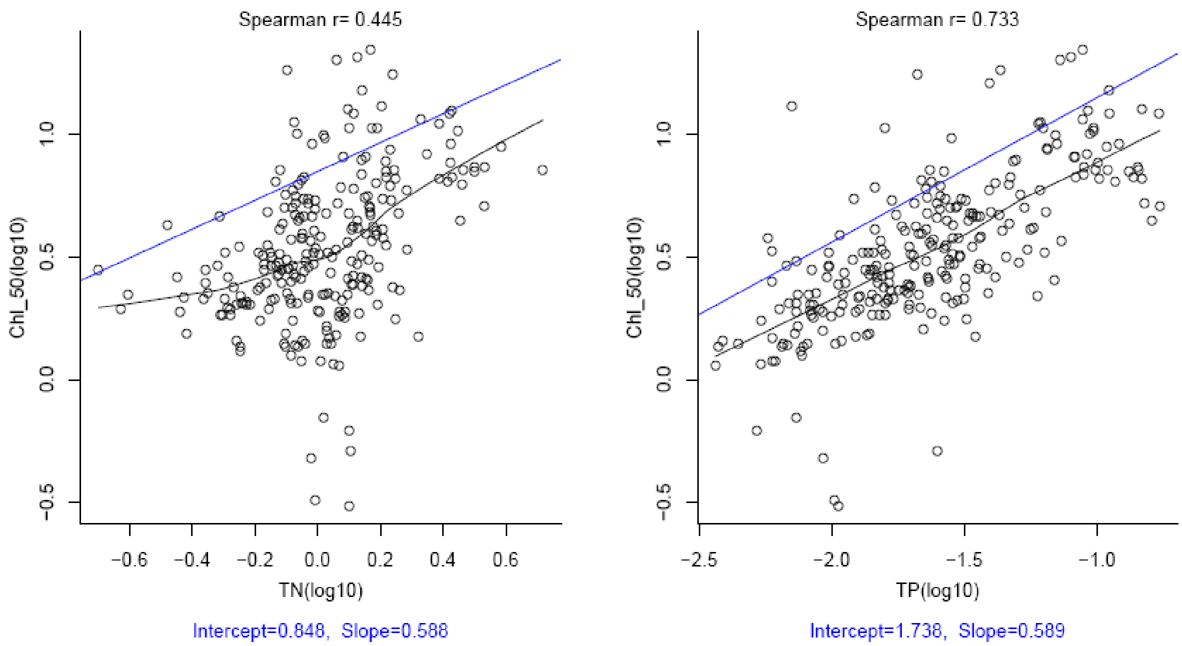

Figure 5-15. Bivariate plot of annual geometric mean chlorophyll *a* as a function of TN and TP across all regions together with Spearman's rho correlation coefficient, LOESS smoother (black line, f=0.7), and quantile regression results (blue line and blue text)

5.5 Marine Waters: Classification and Data Sources

The watershed-based approach for identifying estuarine waterbodies described in Chapter 3 is less suited in South Florida due to its highly managed inland flows and open-water dominated systems (i.e., Florida Bay and Keys). Therefore EPA considered an approach to enhance waterbody identification with data from the:

- Southeast Environmental Research Center (SERC) water quality monitoring network

The SERC Water Quality Monitoring Network data includes monthly to quarterly sampling results from more than 350 sites collected from 1992 to 2009.[15] There are more than 24,000 observations each of TN, TP, and chlorophyll *a* in the SERC database. In cooperation with EPA, scientists for the National Park Service and Florida International University used principal component and cluster analysis to segment these waters based on their similarities and differences. The SERC data are unique to South Florida and are appropriate for principal component and cluster analysis because they provide a long-term record with consistent sampling and analytical methods. EPA is considering use of the SERC water quality monitoring data as a primary source for deriving numeric criteria for South Florida's marine waters, and may consider additional data from IWR Run 40, peer-reviewed literature, reports, and other data sources as mentioned in Section 2.4.

[15] These data are included as part of Florida's IWR database or can be accessed from http://serc.fiu.edu/wqmnetwork/.

The overall segmentation approach recognizes that geomorphologic and geochemical characteristics vary throughout a basin and can alter the impact of nitrogen/phosphorus pollution on water quality and resident communities. Based on this recognition, numeric criteria can be derived for regions with consideration that different areas react differently to nitrogen/phosphorus pollution. Overall, the principal component and cluster analysis applied here follows a line of research initiated in the 1990s. The current analysis EPA is considering consolidates the previous work described below, and uses the most recent data.

Florida Bay has a history of water quality regionalization. Researchers provided an early perspective of community distribution in their calculation of SAV productivity and spatial patterning in Florida Bay (Zieman et al. 1989). In their study, Ziemann et al. (1989) visually determined regional differences through aerial images and consequential ground truthing, and subsequently found a correlation between SAV standing crop and biological and water quality parameters. Similarly, Philips and Badylink (1996) evaluated the correlation between water quality and phytoplankton standing crop, noting that ecologically independent regions were dynamic with the changing seasons. The seasonality of nutrient pollution loadings and consequential water quality is also highlighted by Boyer et al. (1999). Philips and Badylink (1996) subsequently modeled the relationship of chlorophyll *a* and nutrient pollution concentrations to conclude the phosphorus is the major limiting nutrient in Florida Bay. Fourquean et al. (1993) also explained the phosphorus limitation of phytoplankton in Florida Bay, using a principal components analysis to identify three key processes that independently controlled the composition of the studied water column: evaporation-driven concentration of dissolved material, delivery of phosphorus through water exchange with the Gulf of Mexico, and delivery of freshwater with nitrogen pollution. Boyer and Briceno (2007) conducted trend analysis to further understand such underlying processes that dictate Florida's coastal water quality.

Key to all of these studies is the approach of sampling many water quality indicator variables to attain a perspective that recognizes the importance of larger, watershed and land use influences on the resulting water quality (Caccia and Boyer 2005). The principal components analysis EPA is considering uses a variety of water quality indicator variables. Researchers found that a variety of water quality attributes, which affect how coastal and estuarine regions respond to nitrogen/phosphorus pollution, should be considered in addition to the pollutant loads themselves (Boyer and Briceno 2007; Boyer and Briceno 2009; Boyer et al. 1997; Boyer et al. 1999; Briceno and Boyer 2010; Caccia and Boyer 2005; Fourqurean et al. 2003; Fourqurean et al. 1993; Frankovich et al. 2010; Phlips and Badylak 1996; Zieman et al. 1989).

Other studies employ statistical modeling to delineate regional boundaries, form relationships, and facilitate habitat prediction based on water quality indicator variables. For example, two researchers (Fourqurean et al. 2003; Frankovich et al. 2010) used hierarchical cluster analysis to separate benthic SAV habitats into ecologically distinct regions, and subsequent discriminant function analysis to predict benthic habitat based on associated water quality indicator variables. Evaluation of the resulting predictions showed that these analyses were able to determine habitat with a high level of accuracy. Boyer et al. (1997) used stationary monitoring sites to collect multivariate data, from which cluster analysis determined similar regions. Comparison of data among the regions allowed establishment of underlying water quality differences. EPA is considering this approach to determine unique regions via statistical clustering and analysis of significance.

The segmentation EPA is considering refers back to several existing frameworks found in the literature. Cosby et al. (2005) discuss updates to the FATHOM salinity model in the Florida Bay to assess the effects of potential management scenarios. The FATHOM model assesses solute fluxes among basins of the Florida Bay, whose limits are defined by a series of naturally-occurring shallow banks. EPA is considering using these existing morphology-based divisions to create GIS-layer polygons from the cluster analysis of Florida Bay.

In the Florida Keys, EPA is considering to use segmentation developed by Klein and Orlando (1994), who considered water circulation determined by local bathymetry, meteorology, and hydrology in nearshore environments to divide the Florida Keys National Marine Sanctuary into 9 sub-regions. These boundaries, which represent transitions between areas with different transport activity, are dynamic due to seasonal and other climate changes. The adoption of these localized segmentations are supplemented by establishing contour lines between water sampling stations following benthic depth contours, collectively providing division of South Florida's estuaries and coastal basins and a basis that EPA is considering for numeric criteria assignment. A preliminary, summary-level report detailing the proposed segmentation results for South Florida is provided as Appendix D. Figure 5-16 shows the proposed segmentation plan provided by these researchers in comparison to the current estuarine and coastal FDEP-defined WBIDs.

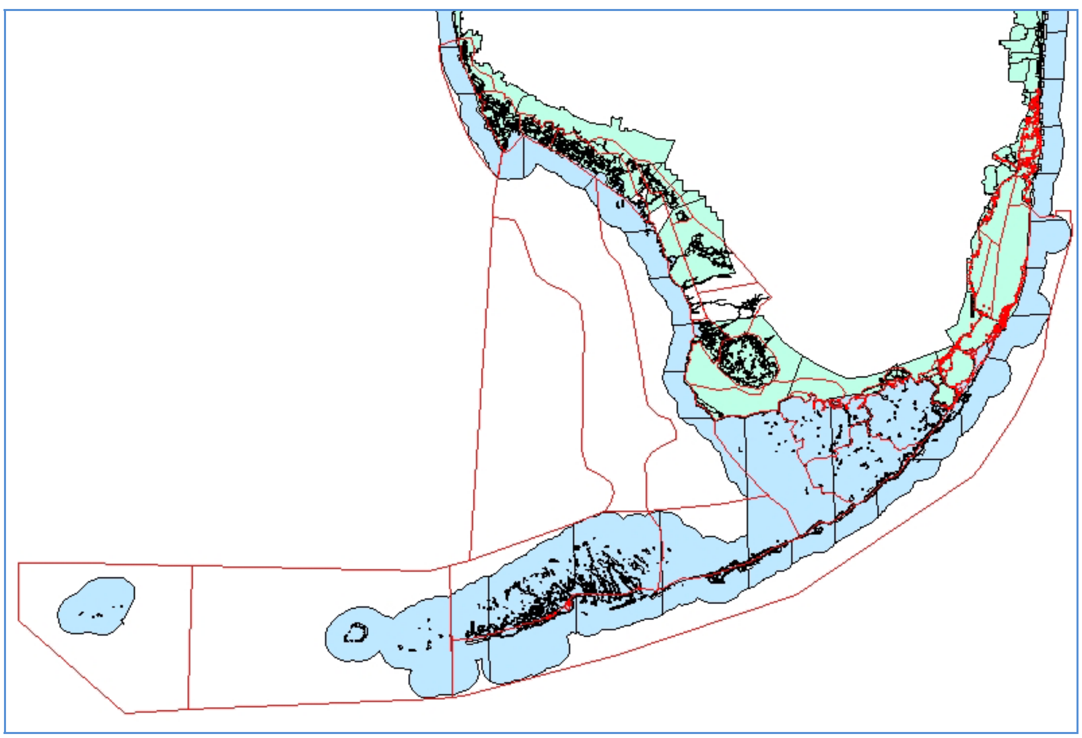

Figure 5-16. Proposed segmentation (red lines) for South Florida. (Criteria derived as a result of this document would only be applicable to waters within 3 nautical miles of land [i.e., those waters highlighted in blue or further landward].)

5.6 Marine Waters: Numeric Criteria Derivation

EPA is considering two approaches for deriving criteria using a reference condition approach. One approach is to estimate criteria from a statistical distribution of water quality indicator variable data. Criteria are derived based on an inclusive distribution of nutrient values that are temporally and/or spatially averaged and are obtained from least-disturbed reference sites from a prescribed region. This is the same approach as for inland flowing waters (see Section 5.4.1.1). For marine waters, EPA is also considering a second approach based on the statistical distribution of raw data and evaluated using a binomial test. In this approach, EPA would derive two criteria: 1) an average (median) concentration and 2) an upper percentile concentration. By considering an upper percentile, this second approach would be sensitive to detecting changes in the distribution of higher concentrations that may be "averaged-out" in annual geometric means. The reference condition approach identifies concentrations that are presumed to be inherently protective of the waterbody because those concentrations are associated with demonstrated healthy biological communities.

5.6.1 Reference Condition Approach—Least Disturbed Sites

5.6.1.1 Annual Average Concentration

The approach that would be used to compute criteria in the form of annual geometric means would be the same as Section 5.4.1.1.2. Please see Section 5.4.1.1.2 for a description of this approach.

5.6.1.2 Binomial Test to Maintain Current Conditions

In some situations, it might be ecologically meaningful to specify numeric criteria applicable to individual observations rather than annual statistics. For example, short duration increases in chlorophyll that are expected during short periods of the summer, might not be well represented in criteria solely based on central tendency measures.

EPA is considering an approach that provides a two-number criterion that would be evaluated using a binomial test. The two-number threshold is particularly applicable to an indicator variable such as chlorophyll where changes may be gradual, but where there is also a desire to avoid infrequent blooms.

EPA is considering the derivation of a two-number criterion ($C_{0.50}$ and $C_{0.25}$) that should not be exceeded by more than 50 percent and 25 percent of the samples at the 90 percent confidence level over an assessment period of three years using the binomial test. EPA may also consider alternative percent exceedances and confidence levels.

EPA is considering this formulation of the binomial approach because it (1) provides a quantitative evaluation of the probability that the data collected during the assessment period are from the same or nearly the same distribution as the observations during the baseline period; (2) the approach is resistant to influence of extremely high values; (3) the approach does not require or imply an assumption of normality or log normality; (4) the approach is not affected by, or is minimally affected by, censored data, and; (5) the approach tests for changes in both the central tendency and the upper end of the concentration distribution.

Under the assumption that the water body fully supports aquatic life use, one method to computing the criteria magnitudes is to set $C_{0.50}$ and $C_{0.25}$ to the maximum 50th and 75th percentile concentration for any 3-year period from the supporting data set.

5.6.1.3 Example: Binomial Test to Maintain Current Conditions

As discussed in Section 5.6.1.2, EPA is considering the derivation of numeric criteria that would be evaluated with a binomial test using individual observations. To illustrate this approach, data from the SERC water quality monitoring network for the South Central Outer Bay Segment of Biscayne Bay are used. This data set includes 11 sites and 14 years (1995-2008) of data. Figure 5-17 displays TN, TP, and chlorophyll *a* boxplots as a function of year and site. TN median ranges from 0.12 mg/L in 2002 and 0.17 mg/L at Marker G-1B to 0.28 mg/L in 2003 and 2004 and 0.30 mg/L at Totten Key. TP median ranges from 0.003 mg/L in 2004 and 0.0046 mg/L at Midbay South to 0.008 mg/L in 2000 and 0.0057 mg/L at Biscayne Channel. Chlorophyll *a* median ranges from 0.16 µg/L in 1999 and 0.18 µg/L at Featherbed Bank, to 0.36 µg/L in 2003 and 0.34 µg/L at Biscayne Channel.

Although different percentiles may be considered, the example application in this section is based on deriving criteria that would be used to statistically test whether more than 50 percent or 25 percent of the results exceed the criteria over a three year period. One method is to set the criteria to the to the maximum 50th and 75th percentile concentration for any 3-year period from the supporting data set. These results are presented graphically in Figure 5-18 for TN, Figure 5-19 for TP, and Figure 5-20 for chlorophyll *a*.

Under the assumption that the water body fully supports aquatic life use, the null and alternate hypotheses are

$$H_o : p \leq p_o$$
$$H_a : p > p_o$$

where p is the fraction of data exceeding the criteria during the assessment period and p_o is the fraction of data exceeding the criteria in its derivation (i.e., 0.50 or 0.25).

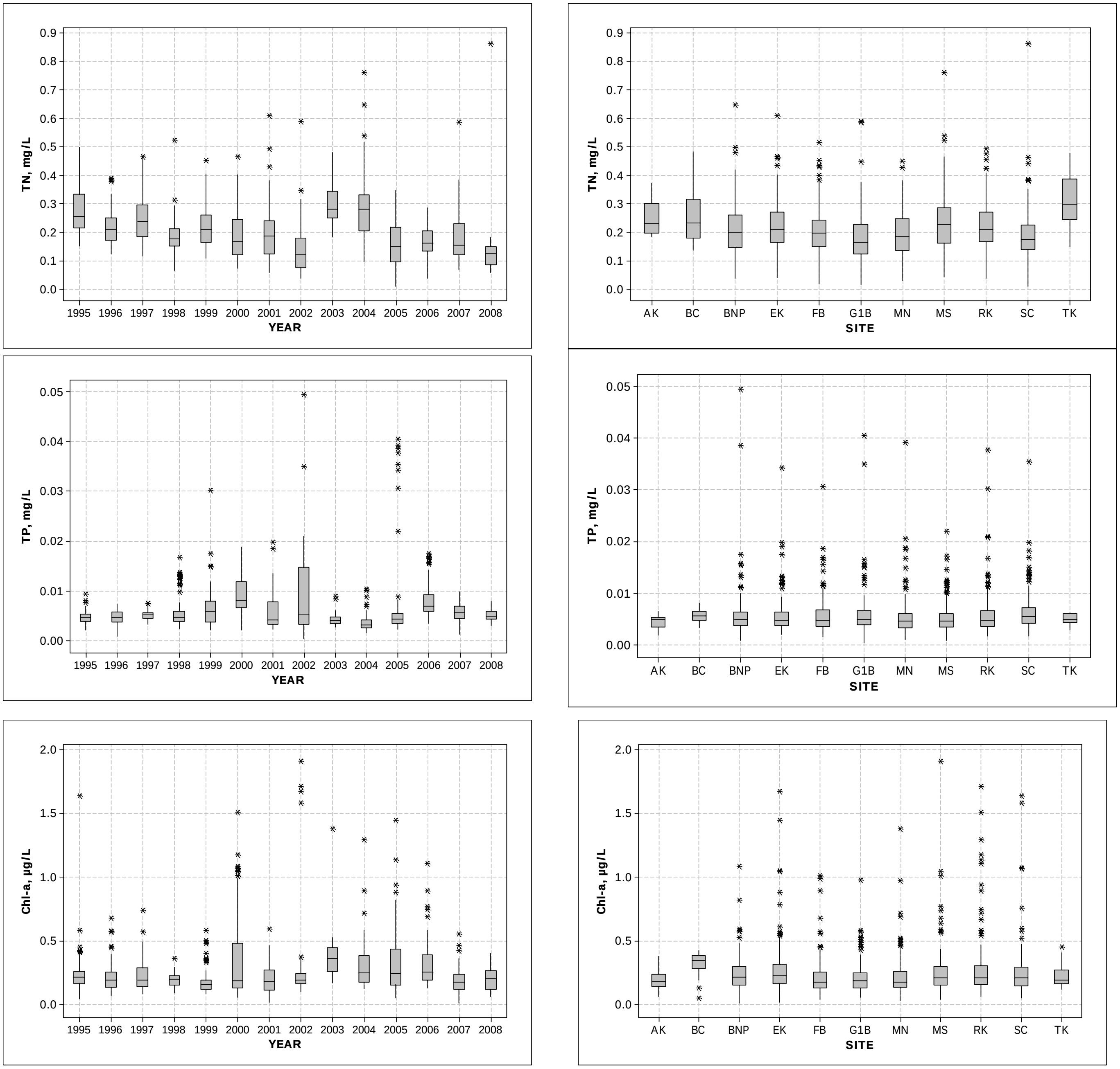

Figure 5-17. Boxplots of TN, TP, and Chl-a for Biscayne Bay's South Central Outer Bay using SERC water quality monitoring network data from 1995-2008 by year and site. (Site key: Adams Key (AK), Biscayne Channel (BC), BNP Marker C (BNP), Elliott Key (EK), Featherbed Bank (FB), Marker G-1B (G1B), Midbay North (MN), Midbay South (MS), Rubicon Keys (RK), Sands Cut (SC), Totten Key (TK))

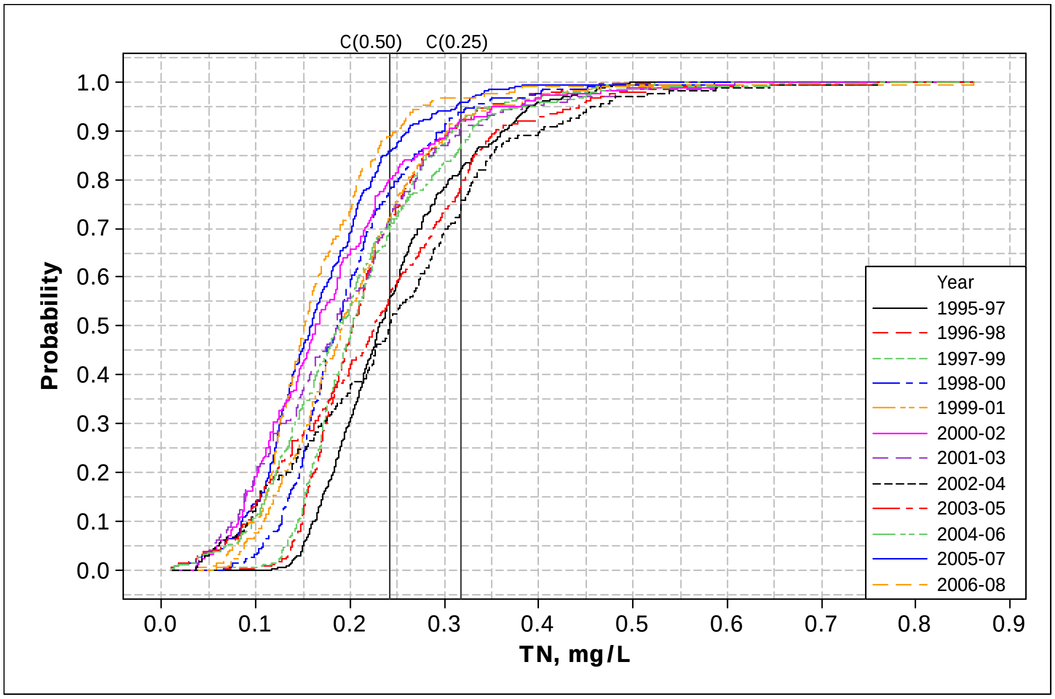

Figure 5-18. TN CDF for Biscayne Bay's South Central Outer Bay using SERC water quality monitoring network data from 1995-2008 by 3-year rolling window

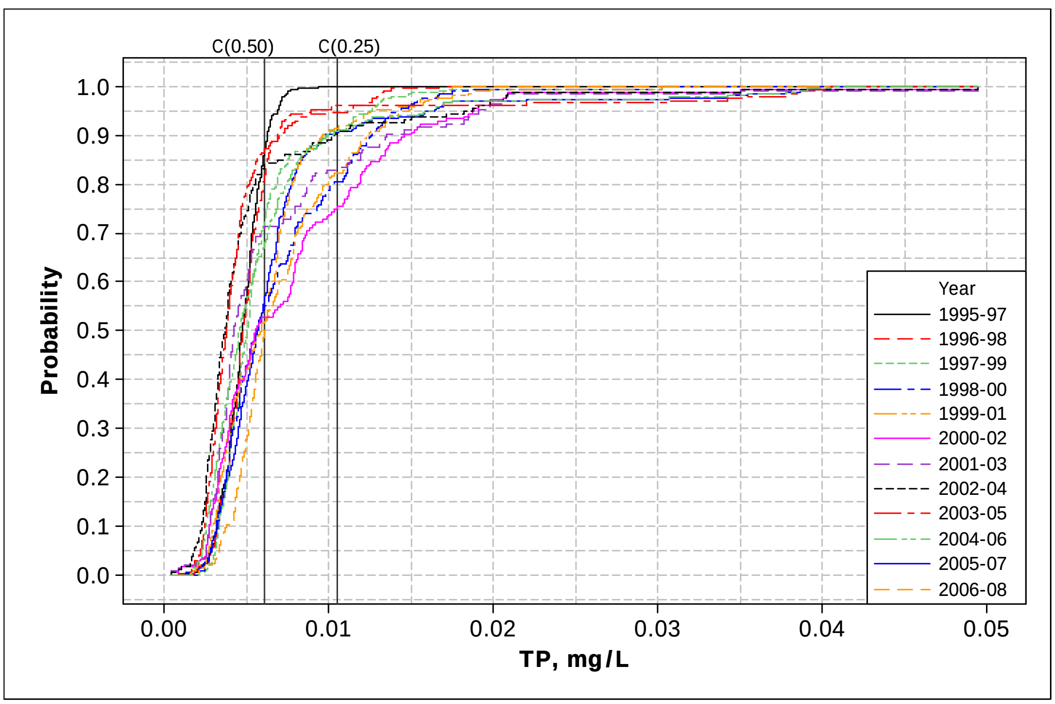

Figure 5-19. TP CDF for Biscayne Bay's South Central Outer Bay using SERC water quality monitoring network data from 1995–2008 by 3-year rolling window

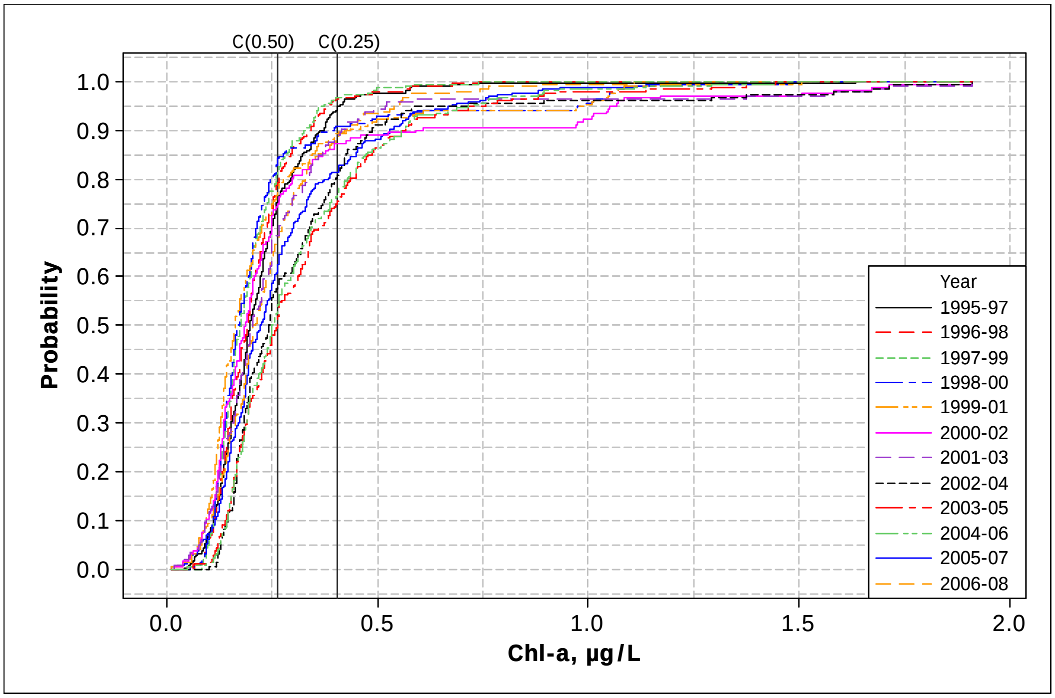

Figure 5-20. Chlorophyll-a CDF for Biscayne Bay's South Central Outer Bay using SERC water quality monitoring network data from 1995–2008 by 3-year rolling window

5.7 Uncertainties and Data Gaps

Deriving numeric criteria for South Florida inland flowing waters presents technical challenges due to the managed flows of canals that are the majority of flowing waters in this region, as well as the generally complex hydrogeology in the region. For example, soil type, particularly in the EAA, where the soils have been physically and chemically modified (e.g., fertilized, oxidized), can influence inland flowing waters in South Florida. Distinct differences in soil types may justify classification of flowing waters by soil type in South Florida. EPA's proposed approach currently considers up to five subregions to address this issue; however, EPA will need to evaluate the advantages and disadvantages to deriving numeric criteria for five subregions separately vs. for South Florida as a whole.

Reference conditions may be developed based on historical data for the system of interest or by comparison with data for other systems. Use of a historical reference condition is based on the availability of sufficient data to document ecological conditions and/or nitrogen and phosphorus concentrations during the reference period. Use of inter-system comparison to develop numeric criteria is based on the availability of data for a sufficient number of comparison systems.

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6 Numeric Nutrient Criteria Development for the Protection of Downstream Estuaries

Pursuant to 40 CFR 131.10(b), water quality standards must ensure the attainment and maintenance of downstream water quality standards. Thus, EPA is deriving numeric criteria for streams in Florida in order to protect the estuarine waterbodies that ultimately receive nitrogen/phosphorus pollution from the watershed. These criteria, which EPA will refer to as Downstream Protection Values, or DPVs, will apply in place of the stream's TN and TP criteria if the applicable DPV is more stringent.

40 CFR 131.10(b)
In designating uses of a water body and the appropriate criteria for those uses, the State shall take into consideration the water quality standards of downstream waters and shall ensure that its water quality standards provide for the attainment and maintenance of the water quality standards of downstream waters.

The DPV criteria will be computed such that the TN and TP discharged from a stream, after accounting for any expected losses during transport, will not contribute a disproportionate fraction of the maximum TN or TP loading protective of water quality standards in the estuarine receiving water. The proportionate fraction will be based on the fraction of total freshwater flow contributed by the reach. Because of the complexities associated with the managed flows in South Florida inland flowing waters (Chapter 5), the fraction of TN or TP from the upstream tributary reach that eventually reaches the estuary cannot be estimated or predicted. Therefore, EPA is considering expressing DPVs at the terminal reach of the tributary into an estuary as protective concentrations or, alternatively, protective loads.

6.1 Analysis Plan

The approach that EPA is considering for developing stream DPV criteria will begin with estimates of limits on TN and TP loading rates that are needed to support balanced natural populations of aquatic flora and fauna in estuarine waters (Figure 6-1). The loading limits will be determined as part of the criteria development effort for estuarine waters as described in Chapter 3 of this document. The protective load limits can be scaled by average streamflow entering the estuary to determine numeric criteria for TN and TP concentrations in streams as they discharge into estuaries. These segments or "reaches" of streams and rivers are referred to as "terminal reaches." Finally, DPVs can be determined for upstream reaches within watersheds by accounting for expected loss or permanent retention of TN and TP within the stream network. The fraction of TN or TP transported in a reach that ultimately reaches estuarine waters is referred to as "fraction delivered."

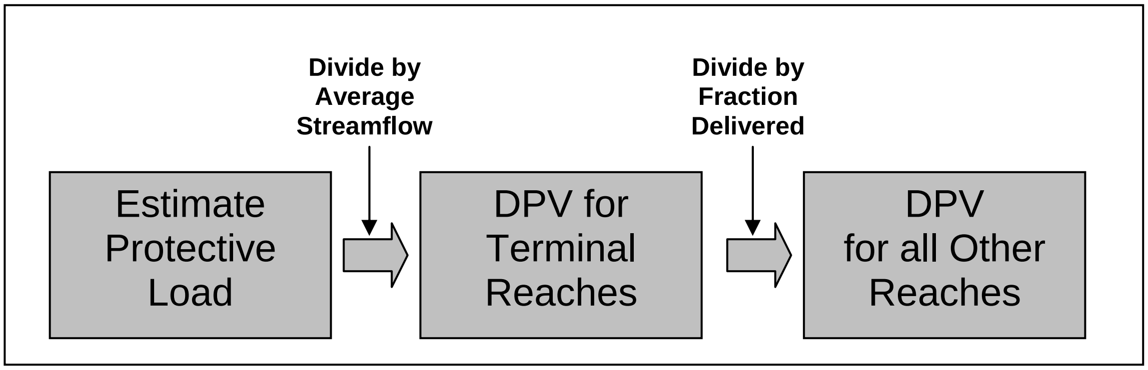

Figure 6-1. The major steps involved in development of numeric criteria for TN and TP in streams and rivers protective of water quality standards in downstream estuaries.

6.2 Estimating Protective TN and TP Loading

EPA is considering using the estimates of protective nitrogen and phosphorus concentrations or loading rates that result from application of its approach for estuarine criteria development, described in Chapter 3 of this document. Although one may estimate a maximum loading rate to an entire estuary, it is possible that different regions within an estuary may be more or less sensitive to nitrogen/phosphorus pollution loading. These differences could affect the TN and TP concentrations that would be needed in streams to protect downstream estuarine waters. For example, water quality responses to nutrient loading could be less in a segment of an estuary that exchanges water freely with the coastal ocean than in another segment that is relatively more isolated from such exchanges. Thus, EPA will consider for each estuary whether it is appropriate to establish separate protective TN and TP loading rates applicable to different estuarine segments. Coupled hydrodynamic-water quality models, segment-specific regression models, or similar approaches may be used to develop estimates for the protective loads (Chapter 3). In the absence of information regarding the differences in the sensitivity of estuarine segments to nitrogen/phosphorus pollution loading, EPA will compute a single protective loading rate for the entire estuary.

6.3 Computing DPVs for Terminal Reaches

In this approach, DPVs for terminal reaches could be computed by dividing the average protective loading rate for an estuary or estuary segment by the sum of the average total streamflow from all stream reaches discharging into the estuary or segment (Equation 6-1).

$$\overline{C_T} = \frac{\overline{L}}{\overline{Q}} \qquad \text{Equation 6-1}$$

where $\overline{C_T}$ is average concentration specified as the terminal reach DPV, $\overline{L}$ is the average loading rate determined to be protective of designated use in the receiving water, and $\overline{Q}$ is the average freshwater inflow to the receiving water body or segment. For the purpose of computing DPVs, the estimate of the average protective loading would exclude loads resulting from direct atmospheric deposition to estuarine surface waters and point-source loads discharged directly to estuarine waters. Similarly, the estimate of total streamflow would not include freshwater inputs resulting from net deposition onto the surface of the estuary (i.e., precipitation minus evaporation) and point source discharges of freshwater into the estuary.

Estimates of average streamflow would be obtained from a watershed model because streamflow is not monitored in many streams. Moreover, if monitoring of streamflow occurs at some distance upstream from the point of discharge into the estuary, it would not represent the fraction of flow resulting from the ungauged portion of the watershed. EPA is considering using the LSPC watershed model for this purpose (Loading Simulation Program in C++; http://www.epa.gov/athens/wwqtsc/html/lspc.html). LSPC would be implemented on a streamflow network derived from the National Hydrography Dataset Plus, or NHDPlus (http://www.horizon-systems.com/nhdplus/index.php). NHDPlus watersheds and flow-lines will be abstracted or "dissolved" to achieve a stream network at approximately the 12-digit HUC scale. This approach would provide improved accuracy and spatial resolution afforded by NHDPlus yet reduce the computational demand by reducing the number of simulated sub-watersheds and stream reaches relative to NHDPlus. LSPC model runs would simulate daily average streamflow for 1997–2009. EPA intends to evaluate the performance of the model for simulating long-term average streamflow, seasonal average streamflow, high flows, low flows, and storm hydrographs using time series of monitored flows at local monitoring stations. EPA also expects to calibrate and evaluate LSPC in order to predict observed stream velocity. Additional information regarding EPA's proposed implementation of LSPC is included in Sections 3.3.3.1 and 3.3.4.

6.4 Computing DPVs for Upstream Reaches

In this approach, DPVs for upstream reaches would be computed from DPVs for downstream terminal reaches via

$$\overline{C}_i = \frac{\overline{C}_T}{\overline{F}_i} \qquad \text{Equation 6-2}$$

where $\overline{C}_i$ is the DPV for an upstream reach and $\overline{F}_i$ is the average fraction of TN or TP transported out of that reach that eventually enters the estuarine receiving water, also called average "fraction-delivered." EPA would compute F_i at a daily time interval ($F_{t,i}$) for each reach, then compute $\overline{F}_i$ as the long term average. Daily values would be computed using

$$F_{t,i} = \prod_j e^{-k_{t,j} t_{t,j}} \qquad \text{Equation 6-3}$$

where $F_{t,i}$ is the fraction-delivered for TN or TP on day t from sub-watershed i. The values of j specify the sequence of stream reaches that comprise the flow path to estuarine waters from reach i. The value $k_{t,j}$ and $t_{t,j}$ are first-order decay rates and reach time-of-travel, respectively, on day t for reach j. EPA is considering using a flow network based on NHDPlus hydrology and daily estimates of stream velocity computed using LSPC watershed models to compute daily time of travel for each stream reach.

First-order decay refers to exponential decay where the instantaneous loss rate does not depend on the concentration of TN or TP. Permanent losses of TN in streams are generally attributed to denitrification in stream sediments (Alexander et al. 2009). Therefore, loss rates for TN are affected by stream attributes that determine the rate that stream water encounters stream sediments. For example, TN losses are inversely associated with stream depth and mean streamflow, the latter of which is likely correlated with stream depth (Alexander et al. 2009; Böhlke et al. 2009). First-order decay rates k_{TN} and k_{TP} generally cannot be measured directly at the scale of whole watersheds and therefore are often estimated empirically. SPARROW watershed models at both regional and national scales are recognized as useful tools for obtaining these estimates. The south Atlantic, Gulf and Tenneessee (SAGT) regional SPARROW model estimated k_{TN} separately for stream reaches in several stream categories based on the magnitude of flow (Table 6-1) (Hoos and McMahon 2009). EPA is considering computing F_i (Equation 6-3) using estimates of flow-dependent k_{TN} from SAGT-SPARROW. EPA is also considering an alternative and slightly simpler approach in which only a single value is used for k_{TN} (e.g., k_{TN}=0.14 d^{-1}), representing a stream of intermediate size (Alexander et al. 2000; Böhlke et al. 2009). This simplification may be justified by the large range in measured stream N loss rates, even for a stream of a given size or depth (Alexander et al. 2009), Regardless of the approach used to determine k_{TN}, estimates of F_i would depend on stream reach time of travel and potentially stream flow obtained from the LSPC watershed model.

Table 6-1. Empirical estimates of the first-order loss rates for TN in southeast U.S. streams from two formulations (Model A and Model B) of the SAGT-SPARROW regional watershed model (Hoos and McMahon 2009)

Average Stream Flow (m^3/s)	Model A k_{TN} (d^{-1})	Average Stream Flow (m^3/s)	Model B k_{TN} (d^{-1})
Q > 28	0.00	Q > 28	0.014
2.8<Q<28	0.13	Q < 28	0.14
Q < 2.8	0.23		

Loss rates for TP represent permanent retention (i.e., burial or binding in sediments) since there are no biogeochemical processes for P comparable to denitrification, which transforms reactive N species (e.g., NO_3^-) into N_2 gas. Permanent P retention in streams is expected to be smaller in streams than N losses, but potentially significant in lakes and especially deeper reservoirs, where sediment-associated P may accumulate more readily. Garćia et al. (2010) model the rate of P retention in southeast U.S. streams as a first-order loss rate inversely proportional to average stream depth, empirically determining k_{TP} (d^{-1}) = 0.049/z, where z is mean stream depth (m). This approach follows the formulation of Alexander et al. (2008), who addressed N and P transport in the Mississippi River basin. Similar to TN, EPA is evaluating whether to use estimates of first-order TP loss rates (i.e., retention for TP) for TP in southeastern U.S. streams estimated using a SPARROW watershed model (Garćia et al. 2010) or, alternatively, a constant value of k_{TP}=0.014 d^{-1}. In order to implement the SPARROW-based estimates of k_{TP}, EPA will need estimates of average stream depth, which EPA is considering obtaining at a daily time step from the LSPC watershed models. Estimates of kTP would be scaled up to compute Fi in the same manner as for TN: estimates of k_{TP} would be used in Equation 6-3 to compute F_i with daily estimates of time-of-travel computed via LSPC watershed model simulations. Average F_i would be used in Equation 6-2 compute DPVs for TP.

6.5 Computing DPVs for South Florida Marine Waters

To address the protection of marine waters downstream from South Florida inland flowing waters, EPA is considering a range of approaches. First, EPA acknowledges the existing numeric criterion for TP in the Everglades Protection Area (10 ppb), and thus has described its approach to ensure that flowing waters upstream of this area assure the maintenance of that criterion (see Section 6.5.2). Second, EPA is considering derivation of protective TN and TP values for the marine receiving waters in South Florida based on the South Florida marine waters criteria (see Chapter 5). These values may take the form of either protective concentrations, or loads from tributaries entering the downstream receiving water, based on existing models or measurements. This approach diverges from those described above for inland flowing waters outside of South Florida because the fraction of TN or TP from the upstream tributary reach that eventually reaches the estuary cannot be estimated or predicted in South Florida due to the region's altered and managed hydrology. For this reason USGS modelers excluded South Florida from their SAGT SPARROW model (Hoos and McMahon 2009). For the same reason, EPA is considering the assignment of DPVs for South Florida at the terminal reach and not at locations farther upstream.

In South Florida a protective load, as an alternative to concentration, may be considered as a way to express the DPV. One advantage of this approach is that loads may be more directly related to impacts on receiving waters and how the waters are managed during the dry and wet seasons. One advantage of expressing DPVs as concentrations is that in upstream reaches a concentration can be directly compared to a stream IPV, and an assessment can be made as to whether the water quality standard is simultaneously being protective of conditions both instream and downstream. A disadvantage of applying DPVs only at terminal reaches is that this approach does not provide an ability to simultaneously assess downstream protection at both upstream and downstream locations. However a downstream estuary would be protected by this approach.

EPA is considering several strategies for calculating DPVs at a tributary terminal reach. There are two steps involved in deriving DPVs: calculating the nitrogen/phosphorus pollution load that protects the estuary, and translating this load into DPVs for contributing waters. In South Florida, the DPV may be expressed either as a load from, or a concentration at, the terminal reach of each tributary to the estuary.

The assignment or allocation of DPVs to multiple tributaries which may be influenced by different combinations of point and non-point sources can be based on an equitable strategy. One example is to assign loads or flow-weighted concentrations to each tributary based on its relative contribution to the estuary. For example, a flow-weighted concentration is calculated as

$$C = \frac{c_1 q_1 + c_2 q_2 + c_3 q_3 + \ldots + c_n q_n}{q_1 + q_2 + q_3 + \ldots + q_n} \qquad \text{Equation 6-4}$$

where C is the protective total influent concentration (DPV) for the estuary over the time period of interest, c_i and q_i are the concentration and flow respectively in tributary "i", and n is the number of such tributaries. If there is more than one tributary (i.e., n>1), then there is no unique solution to Equation 6-4. In this case, achievement of the DPV may be attained through various combinations of individual tributary concentrations. Robust datasets of water quality and flow for each tributary are necessary in order to compute acceptable individual tributary concentrations this way.

Tributary loads can be apportioned most simply according to flows as follows

$$L_i = T\left(\frac{q_i}{q_t}\right) \qquad \text{Equation 6-5}$$

where L_i is the acceptable load from tributary "i" over the time period of interest, q_i is the tributary's flow over the same time interval, q_T is the summed flow to the estuary from all such tributaries, and T is the total load that the estuary can receive and still meet standards. Calculation of loads in this manner implies identical concentrations in all tributaries.

Alternatively, loads for individual tributaries can be calculated based simply on the acceptable summed load to the estuary, irrespective of individual tributary flow considerations. Similar to that described for the Mississippi River Basin and Northern Gulf of Mexico (National Research Council 2009), a simple load allocation equation can be written as

$$L_1 + L_2 + L_3 + \ldots + L_n = T \qquad \text{Equation 6-6}$$

where L_i is tributary "i" load over the time period of interest. Because each tributary drains a different area with a different combination of land uses, pollutant sources, nutrient management practices etc., it is perhaps unrealistic to expect the uniform effluent concentrations that Equation 6-5 implies. The approach described in Equation 6-6 allows more flexibility in DPV setting, by making the total delivered load to the estuary the only constraint.

Potential models that could inform derivation of a protective load include those of Marshall et al. (2008), who provided mass balance calculations of salinity response to freshwater inflows and nutrients in central and south regions of Biscayne Bay, and Cosby et al. (2005), who characterized the hydrologic influences to and water budgets for Florida Bay.

6.5.1 Binomial Approach for Downstream Protection

An alternative approach to that described in the previous section would be to estimate concentrations based on existing monitoring data at terminal reaches of tributaries into the estuaries. This approach would be most applicable in situations where the downstream estuary is currently supporting natural populations of aquatic flora and fauna, and the objective is to ensure that nitrogen and phosphorus concentrations in the tributaries do not increase in order to maintain water quality. For situations where EPA is considering setting criteria to maintain current concentrations, EPA is considering developing a two-number criterion based on the binomial distribution (see Section 3.3.2 for further details on this approach).

6.5.2 Calculating EAA Downstream Protective Values Based on EvPA Total Phosphorus Criteria

The State of Florida has adopted, and EPA has approved, a long-term geometric mean TP criterion for the EvPA of 10 ppb. This 10 ppb long-term geometric mean criterion applies throughout the EvPA (see Section 5.2.1 for more information). In order to ensure the attainment and maintenance of downstream water quality standards, EPA is considering the derivation of numeric criteria for TP within the EAA that are supportive of balanced natural populations of aquatic flora and fauna within the EAA as well as supportive of the current water quality standards (with a TP criterion of 10 ppb) in the EvPA. To do this, EPA could consider historical water quality and pumping records to characterize nutrient attenuation that would, in turn, inform derivation of DPVs for flowing waters in the EAA.

6.6 Specifying Frequency-Duration for DPVs

DPV criteria are intended to limit average nitrogen/phosphorus pollution loading rates from watersheds to estuaries. Although water quality in estuaries has been shown to change in relatively short periods in response to changes in nitrogen/phosphorus pollution inputs, key assessment endpoints, such as the depth of colonization of seagrasses, respond relatively slowly and cumulatively to sustained patterns of increasing or decreasing nitrogen/phosphorus pollution loading (e.g., Greening and Janicki 2006). Moreover, controls on average nitrogen and phosphorus concentration and loading at the scale of whole estuarine watersheds tend to change slowly in response to cumulative effects of multiple anthropogenic causes, whether beneficial or detrimental. These observations suggest that DPV criteria should be structured to reflect this longer time-scale of impact.

Because DPV criteria would be derived from estimates of protective load limits, it is also important to consider the potential impact of systematic bias. EPA will evaluate whether it will be practical to specify DPV criteria as flow-weighted concentrations, or as concentrations that have been empirically adjusted for season and flow-conditions. If not, EPA will specify an alternative approach, potentially simply annual means. EPA is considering the use of arithmetic means for this purpose because although geometric means have often been recommended to quantify central tendency for log-normally distributed data (TN and TP concentrations are often log-normally distributed) the product of geometric mean concentration and average streamflow may not provide reliable estimates of the transport of TN and TP (e.g., Cohn et al. 1989).

6.7 Example Application: Pensacola Bay Watershed

We illustrate the methodology that EPA is considering by providing an example of the computation of DPVs for Pensacola Bay using the LSPC watershed model. The Pensacola Bay system is an estuarine complex located in Florida's panhandle region. It includes several interconnected estuarine bays and sounds and has a large watershed that extends well into the state of Alabama (Figure 6-2 and Figure 6-3). The LSPC watershed model for Pensacola Bay includes 224 sub-watersheds and associated stream reaches, 12 of which are terminal reaches (i.e., discharge directly into the Pensacola Bay; Figure 6-2, Figure 6-3).

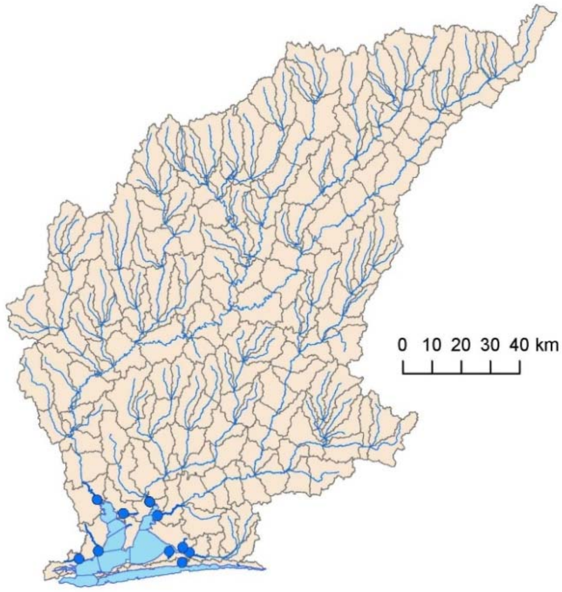

Figure 6-2. Map of Pensacola Bay watershed illustrating the 224 sub-watersheds and associated stream reaches included in the LSPC watershed model. Ten of the twelve terminal reaches discharging into Pensacola Bay are shown with blue dots.

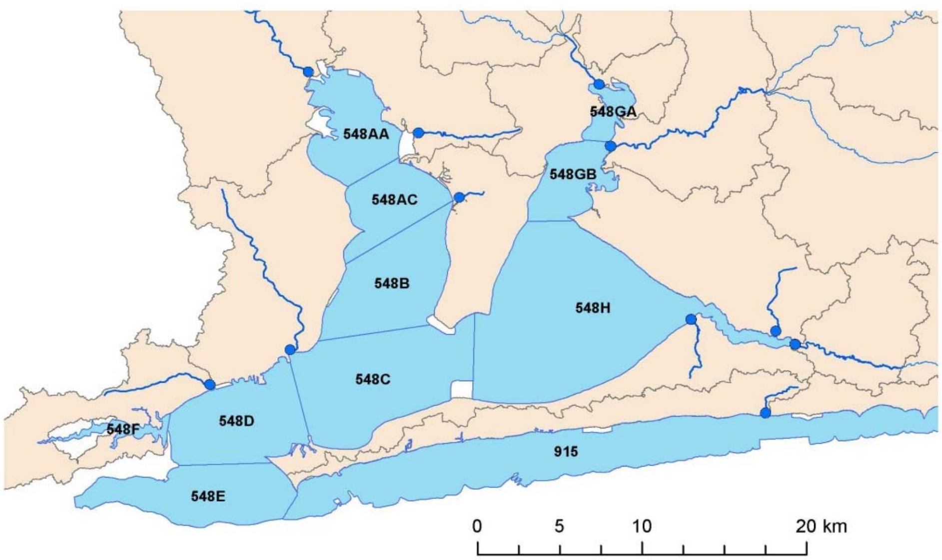

Figure 6-3. Map of Pensacola Bay illustrating where 11 of 12 terminal reaches from the LSPC watershed model enter the bay (not shown is the terminal reach at the eastern end of Santa Rosa Sound). Labels indicate the WBIDs that make up the larger open water areas of Pensacola Bay.

6.7.1 Protective TN and TP Loading and Terminal Reach DPVs

Because protective loading rates for TN and TP have not yet been determined for Pensacola Bay, example calculations are presented here for illustrative purposes only. In this example, the long term (1997–2009) average TN and TP loads to Pensacola Bay were estimated from LSPC model simulations and used as an estimate of the protective loads. The long term average load and average streamflow was used to derive the associated estimates of 0.41 mg/L TN and 0.015 mg/L TP for terminal reach DPVs for the entire bay (Table 6-2; last row). The concentrations that would result from application of EPA's approaches for developing estuarine TN and TP numeric criteria (Chapter 3) may be entirely different and would replace these values in the final computations.

To illustrate how DPVs could be computed on a segment-specific basis, four regions of the Pensacola Bay system were hypothesized to respond differently to nitrogen/phosphorus pollution (again this is for illustrative purposes only). In this example, slightly lower TN and TP concentrations are assumed to be required to support balanced natural populations of aquatic flora and fauna in the Blackwater-East Bay region, which is further from Pensacola Pass than Escambia Bay and Pensacola Bay proper (Table 6-2). It was assumed, for example, that slightly higher TN and TP concentrations would be appropriate in Santa Rosa Sound because the small watershed area limits overall nitrogen/phosphorus pollution loading to this coastal lagoon segment of the bay.

Table 6-2. Hypothetical values illustrating the computation of segment-specific protective loading rates and associated DPV concentrations for terminal stream reaches. Average flows are 1997–2009 averages computed using LSPC.

Region	TN Load (kg/d)	TP Load (kg/d)	Avg Flow (m^3/s)	DPV-TN (mg/L)	DPV-TP (mg/L)
Escambia Bay	7,165	267	193	0.43	0.016
Pensacola Bay	111	4	3.0	0.43	0.016
Blackwater-East Bay	4,657	173	139	0.39	0.014
Santa Rosa Sound	90	3	2.2	0.47	0.018
Total/Average	12,023	447	337	0.41	0.015

6.7.2 Fraction-Delivered in the Pensacola Bay Watershed

The approach described in section 6.4 was applied to compute the TN and TP fraction-delivered for each reach in the Pensacola Bay watershed. The computations were completed for each day during the 1997–2009 LSPC simulation period. Subsequently, long-term averages were computed. To reduce the total number of numeric criteria values that would be computed, estimates of fraction-delivered were rounded upward to the nearest 0.1. Maps depicting the long-term average TN fraction-delivered for each reach (Figure 6-4) contrast the results obtained when using a constant first-order loss rate of 0.14 d^{-1} (panel A) versus flow-dependent values (panel B) for first-order TN loss based on a SPARROW model B (Table 6-1) (Hoos and McMahon 2009). The estimates obtained using flow-dependent loss rates were almost always higher than those obtained assuming a constant loss rate (Figure 6-5). The average difference in the estimate of TN fraction-delivered was 6 percent and the maximum difference was 17 percent. Differences between the two approaches decreased as the distance and time-of-travel to the coast decreases and all values converged to 1.0 (Figure 6-5). Because the flow-dependent loss rates based on the SPARROW model example cause larger rivers to transport N with lower loss to the coast, the spatial distribution of fraction-delivered in the flow-dependent scenario was different from that obtained with a constant loss. In particular, areas of more efficient N transport extended further into the watershed along major rivers when compared to the other scenario (Figure 6-4). Although the spatial scale is much smaller, this pattern is similar to that which has been described for N transport in the Mississippi/Atchafalaya River Basin, where N is transported with low losses from the upper Midwest to the Gulf of Mexico (Alexander et al. 2008).

If one assumes a constant first-order loss rate of 0.014 d^{-1} for streams in the Pensacola Bay watershed, then all estimates of TP fraction-delivered are greater than 90 percent, indicating negligible TP losses. Alternatively, following the approach of Garćia et al. (2010) gives fraction delivered estimates for TP between 0.53 and 1.0 (Figure 6-6). EPA is continuing to evaluate which are the most defensible parameter values for modeling TP retention in streams as a first-order decay process. An important consideration is that there are few lakes and reservoirs in the Pensacola Bay watershed. Even though there is at least one reservoir within the watershed, Gantt Lake, located on the Conecuh River, the model was able to adequately simulate hydrology and water quality in the basin without including it. This may not be the case in other watersheds in Florida, where the presence of multiple lakes and reservoirs in the watershed may have more of an effect on TN and TP transport to the coast.

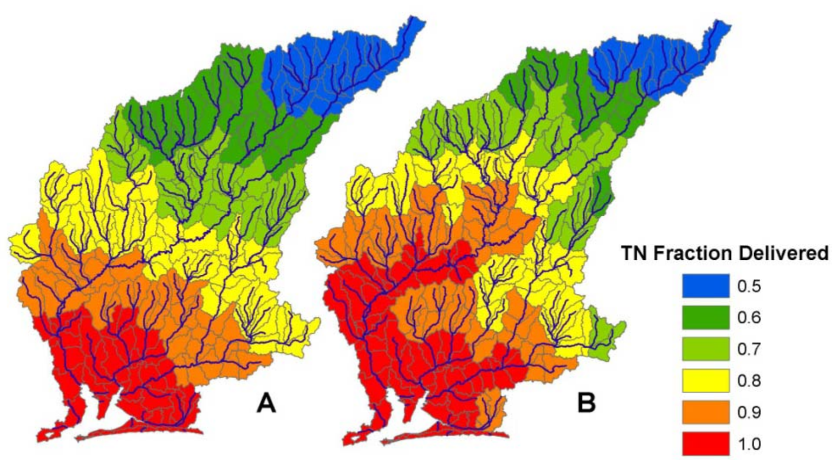

Figure 6-4. Map of average TN fraction-delivered to Pensacola Bay calculated daily for 1997–2009 using (A) a constant first-order decay rate of 0.14 d^{-1} and (B) using first-order decay rates for TN specified on the basis of two ranges of flow rate in the reach (Table 6-1), following the Model B SPARROW regression parameters reported for southeast U.S. streams by Hoos and McMahon (2009). In both cases, the estimates of fraction delivered were rounded up to the nearest 0.1.

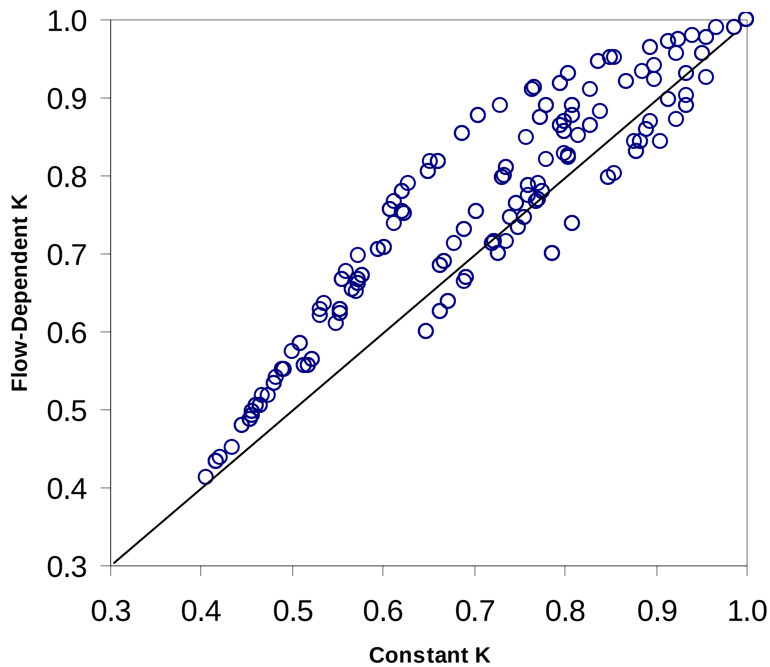

Figure 6-5. The relationship between TN fraction delivered (d^{-1}) in the Pensacola Bay watershed computed using a constant first-order decay rate of 0.14 d^{-1} (horizontal axis) and flow-dependent values for the first-order decay rate for TN, following Hoos and McMahon (2009) (vertical axis)

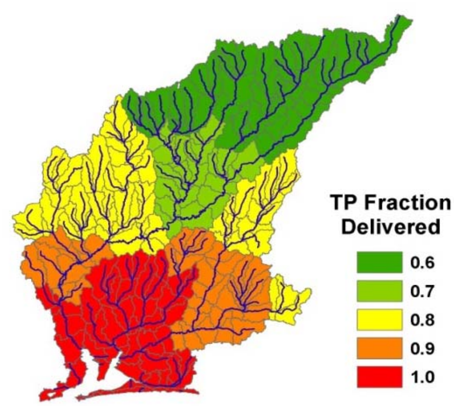

Figure 6-6. A map showing estimates of the TP fraction delivered to estuarine waters in Pensacola Bay as calculated using a TP retention rate (expressed as first-order decay) that is inversely related to stream depth (Garćia et al. 2010)

6.7.3 Computing DPVs for Upstream Stream Reaches

Upstream DPVs were computed following the conceptual approach illustrated in Figure 6-1 and expressed in Equation 6-2 and using the example terminal-reach TN and TP DPV values (Table 6-2). Portions of the watershed draining to each of the respective estuarine regions were determined from the stream network (Figure 6-7, panel A). Using equation 6-2 with estimates of TN fraction-delivered rounded to the nearest tenth, twelve unique TN DPV criteria were determined for corresponding regions of the Pensacola Bay watershed (Figure 6-7, panel B). In several cases, differences between the DPVs were quite small. The same watershed regions correspond to the terminal reach TP criteria (Figure 6-8, panel A). Using the estimates of TP fraction delivered (Figure 6-6) rounded to nearest tenth resulted in nine unique TP DPV values for corresponding regions of the watershed (Figure 6-8, panel B), ranging from 0.014 to 0.027 mg/L TP.

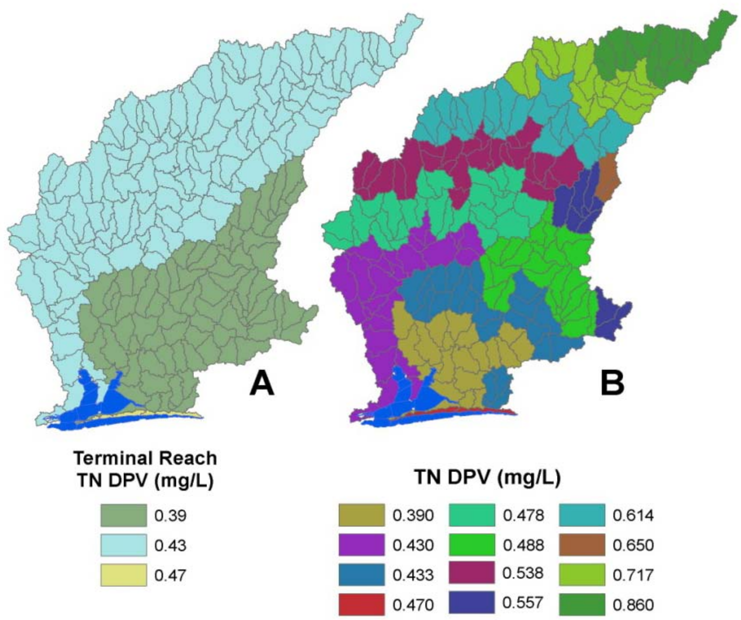

Figure 6-7. Regions of the Pensacola Bay watershed associated with unique terminal reach DPV values for TN (panel A) and the resulting computed values for TN DPVs in all streams (panel B). TN fraction-delivered was rounded up to the nearest 0.1, resulting in 12 unique TN DPV values for the entire watershed. For these maps, fraction-delivered was calculated using stream-flow dependent first-order decay rates for TN, following the results of Hoos and McMahon (2009).

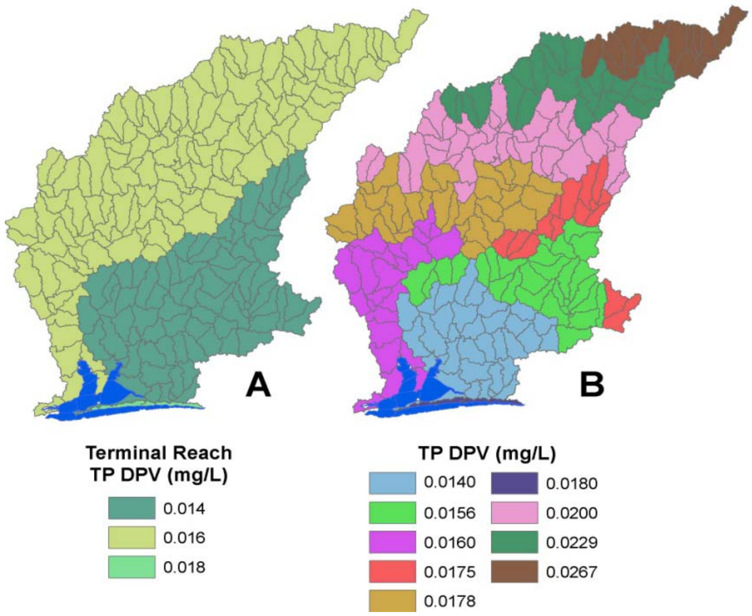

Figure 6-8. Regions of the Pensacola Bay watershed associated with unique terminal reach DPV values for TP (panel A) and the resulting computed values for TP DPVs. Fraction-delivered was rounded up to the nearest 0.1, resulting in nine unique TP DPV values for the entire watershed. For these calculations, TP fraction-delivered was calculated using stream-depth dependent rates following the results of Garćia et al. (2010).

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