Technical Support Document for U.S. EPA's Proposed Rule for Numeric Nutrient Criteria for Florida's Estuaries, Coastal Waters, and South Florida Inland Flowing Waters

Volume 1: Estuaries

November 30, 2012

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Disclaimer

This document supports the U.S. Environmental Protection Agency's (hereafter EPA or the Agency) numeric nutrient criteria proposed on November 30, 2012, pursuant to section 303(c)(4) of the Clean Water Act (CWA) (Title 40 of the *Code of Federal Regulations* [CFR] section 131.43). The information provided herein does not substitute for the CWA or EPA's regulations; nor is this document a regulation itself. Thus, this document cannot and does not impose any legally binding requirements on EPA, states, authorized tribes, the regulated community, or any other party, and might not apply to a particular situation or circumstance.

Contents

Figures

Tables

Abbreviations and Acronyms

%	percent
°C	degrees Celsius
°F	degrees Fahrenheit
ADEM	Alabama Department of Environmental Management
BOD	biochemical oxygen demand
$CaCO_3$	calcium carbonate
CBOD	carbonaceous biochemical oxygen demand
CDOM	colored dissolved organic matter
CE-QUAL-ICM	three-dimensional eutrophication model
CFR	Code of Federal Regulations
cfs	cubic feet per second
chl-a	chlorophyll *a*
CWA	Clean Water Act
DIN	dissolved inorganic nitrogen
DO	dissolved oxygen
DON	dissolved organic nitrogen
DPVs	downstream protective values
DRP	dissolved reactive phosphorus
EFDC	Environmental Fluid Dynamics Code
EPA	United States Environmental Protection Agency
F.A.C.	Florida Administrative Code
FDEP	Florida Department of Environmental Protection
FDNR	Florida Department of Natural Resources
FFWCC	Florida Fish and Wildlife Conservation Commission
FWRI	Fish and Wildlife Research Institute
GAEPD	Georgia Environmental Protection Division
GIS	geographic information system
GLUT	Georgia Land Use TrendsGMFR geometric mean function regression
GSD	geometric standard deviation
GTMNERR	Guana-Tolomato-Matanzas National Estuarine Research Reserve
GTMP	Guana, Tolomato, Matanzas, Pellicer
HAB	harmful algal bloom
HUC12	12-digit hydrologic units
HUC8	8-digit hydrologic units
IWR	Impaired Waters Rule

K. brevis	*Karenia brevis*
K_d	light attenuation coefficient
lbs	pounds
LSPC	Loading Simulation Program in C++
m	meter
m^3/s	cubic meters per second
mg/L	milligrams per liter
mgd	million gallons per day
µg/L	micrograms per liter
µM	micromoles
µmhos	micromhos
NEP	National Estuary Program
NH_3	ammonia
NH_4	ammonium
NHD	National Hydrography Dataset
NO_2	nitrite
NO_3+NO_2	nitrate+nitrite
NO_3	nitrate
NOAA	National Oceanic and Atmospheric Administration
NPDES	National Pollutant Discharge Elimination System
NTU	nephelometric turbidity units
NWFWMD	Northwest Florida Water Management District
OFW	Outstanding Florida Water
PO_4	orthophosphate or phosphate
ppb	parts per billion
ppt	parts per thousand
PCU	platinum-cobalt unit
PLSM	Pollutant Load Simulation Model
PSU	practical salinity unit
RSY	reference segment year
S	normalized sensitivity coefficient
SAV	submerged aquatic vegetation
SBEP	Sarasota Bay Estuary Program
SFWMD	South Florida Water Management District
SJRWMD	St. Johns River Water Management District
SOD	sediment oxygen demand

SRP	soluble reactive phosphorus
SRWMD	Suwannee River Water Management District
SSAC	Site-Specific Alternative Criteria
STORET	STOrage and RETrieval of Water-Related Data
SWFWMD	Southwest Florida Water Management District
SWIM	Surface Water Improvement and Management
TAC	Technical Advisory Committee
TBEP	Tampa Bay Estuary Program
TBNMC	Tampa Bay Nitrogen Management Consortium
TDN	total dissolved nitrogen
TDP	total dissolved phosphorus
TKN	total Kjeldhal nitrogen
TMDL	total maximum daily load
TME	Tolomato–Matanzas Estuary
TN	total nitrogen
TP	total phosphorus
TSD	Technical Support Document
TSI	Trophic State Index
TSS	total suspended solids
U.S.	United States
USFWS	United States Fish and Wildlife Service
USGS	United States Geological Survey
WASP7	Water Quality Analysis Simulation Program Version 7.3
WBID	water body identification number
WSE	water surface elevation
WWTF	wastewater treatment facility
WWTP	wastewater treatment plant
Z_c	depth of colonization

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1. Methods Used to Derive Numeric Nutrient Criteria for Florida Estuaries

This volume presents the process by which EPA derived numeric nutrient criteria for estuaries. EPA derived numeric criteria to protect designated uses for the purposes of the Clean Water Act (CWA) (Section 1.1).[1] EPA selected three nutrient-sensitive biological endpoints and associated water quality targets to derive numeric nutrient criteria for each estuary (Section 1.2). EPA then grouped estuarine waters according to adjacent and similar systems (classification), and further divided each estuary system into segments on the basis of similar biological, chemical, and physical attributes (segmentation) (Section 1.3); the classification serves as an organizing framework for analyses, and the segmentation delineates areas in each estuary where the criteria apply. For each estuary system, numeric nutrient criteria values were derived using one of two approaches, either statistical modeling (stressor-response modeling) or mechanistic modeling (Section 1.4). EPA applied a decision framework to determine, on a segment-specific basis, the final numeric nutrient criteria (Section 1.5). EPA then derived downstream protective values (DPVs) that apply at the points where inland flowing waters flow into estuaries to protect the downstream estuarine water bodies (Section 1.6). Based on the newly-approved State water quality standards that apply to Clearwater Harbor/St. Joseph Sound, Tampa Bay, Sarasota Bay, and Charlotte Harbor/Estero Bay, EPA is not presenting analyses or numeric criteria for these four systems. Since Florida has not developed DPVs for Clearwater Harbor/St. Joseph Sound, Tampa Bay, Sarasota Bay, and Charlotte Harbor/Estero Bay, EPA derived DPVs for these four systems using one of the methodologies presented in Section 1.6. The results of this process—specifically the estuary-specific numeric nutrient criteria, site-specific procedures, and DPVs—are presented in Section 2 using the approaches described in Section 1. The results of analyses related to marine lakes and tidal creeks can be found in Section 3.

1.1. Deriving Numeric Criteria to Protect Designated Uses in Estuarine Waters

Florida's current narrative criterion reads, in part, that "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" [Subsection 62-302.530 (47)(b), F.A.C.]. Given those expectations for state numeric criteria under the CWA and in accordance with the court decision on EPA's December 2010 rule,[2] EPA interpreted FDEP's narrative nutrient criterion of causing no imbalance as equivalent to the prevention of a harmful increase in levels of nutrients.

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[1] On June 13, 2012, FDEP submitted new and revised water quality standards for review by the EPA pursuant to section 303(c) of the CWA. These new and revised water quality standards are set out primarily in Rule 62-302 of the F.A.C. [Surface Water Quality Standards]. FDEP also submitted amendments to Rule 62-303, F.A.C. [Identification of Impaired Surface Waters], which sets out Florida's methodology for assessing whether waters are attaining State water quality standards. On November 29, 2012, EPA approved the provisions of these rules submitted for review that constitute new or revised water quality standards (referred to in this TSD as the "newly-approved State water quality standards").

Among the newly-approved State water quality standards are numeric criteria for nutrients that apply to a set of estuaries and coastal marine waters in Florida. Specifically, these newly-approved State water quality standards apply to Clearwater Harbor/St. Joseph Sound, Tampa Bay, Sarasota Bay, Charlotte Harbor/Estero Bay, Clam Bay, Tidal Cocohatchee River/Ten Thousand Islands, Florida Bay, Florida Keys, and Biscayne Bay. Under the Consent Decree, EPA is relieved of its obligation to propose numeric criteria for these waters.

[2] Case 4:08-cv-00324-RH-WCS, February 18, 2012.

In its decision,[3] the court, in discussing numeric criteria translating Florida's narrative criterion, stated that "the right target was a criterion that would identify a *harmful* increase in a nutrient level – an increase that, in the language of Florida's narrative criterion, would create an 'imbalance' in flora and fauna." Order at 63. Upon review of the latest scientific knowledge, EPA has identified nutrient-sensitive biological endpoints relevant to particular estuarine and coastal systems. EPA determined that maintenance of seagrasses, maintenance of balanced algal populations, and maintenance of aquatic life are three sensitive biological endpoints, which can be measured by water clarity (as it relates to light levels sufficient to maintain historic depth of seagrass colonization), chlorophyll *a* (chl-a), and DO, respectively, and appropriately used in derivation of numeric nutrient criteria that protect the State's designated uses from harmful increases in nitrogen and phosphorus concentrations.

EPA's approach to deriving numeric nutrient criteria is consistent with FDEP's approach to interpreting its narrative nutrient criterion and deriving numeric thresholds at the State level. FDEP has approached the derivation of numeric nutrient criteria in much the same way as EPA by aiming to prevent adverse effects to natural populations of aquatic flora and fauna.[4]

The CWA requires water quality standards, consisting of designated uses of the waters and water quality criteria based on such uses, to restore and maintain the chemical, physical, and biological integrity of the nation's waters. To derive numeric criteria to protect designated uses, it is important to consider the CWA goal of restoring and maintaining biological integrity. This is important because the State's numeric criteria must serve the purposes of the CWA and ultimately restore and maintain water quality integrity. The concept of integrity is commonly defined as "the capability of supporting and maintaining a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of the natural habitat of the region" (Frey 1977). EPA uses an operational definition of integrity as "the ability of an aquatic community to support and maintain a structural and functional performance comparable to the natural habitats of a region" (Frey 1977; Karr and Dudley 1981).

As required by 40 *Code of Federal Regulations* (CFR) 131.11, water quality criteria must be based on sound scientific rationale, must contain sufficient parameters or constituents to protect the designated use, and for multiple use designations, must support the most sensitive use. Because the state has already interpreted what it means to protect designated uses from nutrient pollution through its narrative criterion statement, EPA is proposing to prevent the harmful increase of nutrient levels by ensuring that the numeric values developed will protect nutrient-sensitive aquatic floral and faunal species.

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[3] Case 4:08-cv-00324-RH-WCS, February 18, 2012.

[4] *State of Florida Numeric Nutrient Criteria Development Plan*, Prepared by: Bureau of Assessment and Restoration Support, Division of Environmental Assessment and Restoration, Florida Department of Environmental Protection Tallahassee, FL, March 2009; *Technical Support Document: Development of Numeric Nutrient Criteria for Florida Lakes and Streams*. Florida Department of Environmental Protection, Standards and Assessment Section, June 2009; *Technical Support Document: Development of Numeric Nutrient Criteria for Florida Lakes, Spring Vents and Streams*. Florida Department of Environmental Protection, Standards and Assessment Section, 2012.

1.2. Nutrient-Sensitive Biological Endpoints and Water Quality Targets Used to Derive Numeric Nutrient Criteria

Aquatic systems have, over time diversified populations of aquatic flora and fauna adapted to the environmental conditions in a water body. To develop numeric nutrient criteria that restore and maintain the balance of those populations of species, EPA first determined which biological species or endpoints could be affected by changes in concentrations of nitrogen and phosphorus. EPA also assessed the availability of data that would assist in determining a protective concentration that would prevent an imbalance in populations (i.e., conditions that would cause some species to disappear or some species to dominate, or bloom). More detail regarding EPA's evaluation is in Appendix B of *Methods and Approaches for Deriving Numeric Criteria for Nitrogen/Phosphorus Pollution in Florida's Estuaries, Coastal Waters, and Southern Inland Flowing Waters* (2010 Methods Document) (USEPA 2010a).

Based on its evaluation of the best available scientific literature, EPA used nutrient-sensitive biological endpoints and associated water quality targets to derive numeric nutrient criteria for estuaries. The term biological endpoint is used throughout to describe a floral or faunal component of the environment that the proposed criteria are designed to restore, protect, or maintain. The term water quality target is used to describe the quantitative level of a water quality attribute or constituent that is sufficient to protect the biological endpoint. The term endpoint measure is used to describe the measurement used to determine each water quality target.

EPA is proposing to develop numeric nutrient criteria for Florida's estuarine waters using three biological endpoints that are sensitive to nutrients and necessary to ensure protection of balanced populations of aquatic flora and fauna (see Table 1-1).

(1) Maintenance of seagrasses: Light levels to support historic depth of seagrass colonization will be measured to achieve a segment-specific light attenuation coefficient (K_d) (described as light penetration or water clarity);

(2) Maintenance of balanced algal population: Chl-a concentrations, a surrogate for phytoplankton biomass, will be measured to achieve a chl-a concentration target of 20 µg/L, which must not be exceeded more than 10 percent of the time;

(3) Maintenance of aquatic life: Sufficient dissolved oxygen (DO) concentrations to maintain sensitive aquatic life.

Table 1-1. Terminology to describe endpoints and targets

Biological Endpoint	Endpoint Measure	Water Quality Target(s)	Criteria[a]
Maintenance of seagrasses	Historic depth of seagrass colonization	Segment-specific light attenuation coefficient (K_d) (described as light penetration or water clarity)	Chl-a Total nitrogen (TN) Total phosphorus (TP)
Maintenance of balanced algal population	Chl-a concentrations associated with balanced algal populations (chl-a is a surrogate of phytoplankton biomass)	Chl-a concentrations must not exceed 20 µg/L more than 10 percent of the time	Chl-a TN TP

Biological Endpoint	Endpoint Measure	Water Quality Target(s)	Criteria[a]
Maintenance of aquatic life	Sufficient DO to maintain aquatic life	Minimum allowable DO of 4.0 mg/L as a water column average in an estuary segment 90 percent of the time over the simulation's time span; Daily average DO of 5.0 mg/L as a water column average in an estuary segment 90 percent of the time over the simulation's time span; and Minimum 3-hour average DO of 1.5 mg/L in the bottom two layers of an estuary segment over the simulation's time span[b]	Chl-a TN TP

[a] See Section 2.X.1 and Section 2.X.7 for proposed criteria for specific estuarine segments
[b] The water quality target expressed is an interpretation of Florida's water quality standards in Section 62-302.530, F.A.C.

Table 1-2 shows the endpoints that were evaluated in each of the estuaries.

Table 1-2. Summary of estuary system classification and endpoints

		Endpoints/Targets[a]		
Estuary System		Seagrass	Chl-a	DO
1	Perdido Bay	●	●	●
2	Pensacola Bay	●	●	●
3	Choctawhatchee Bay	●	●	●
4	St. Andrews Bay	●	●	●
5	St. Joseph Bay	●	●	●
6	Apalachicola Bay	●	●	●
7	Alligator Harbor	●	●	●
8	Ochlockonee Bay	●	●	●
9	Big Bend	●	●	●
10	Suwannee Sound	●	●	●
11	Springs Coast	●	●	●
12	Clearwater Harbor/St. Joseph Sound[b]	–	–	–
13	Tampa Bay[b]	–	–	–
14	Sarasota Bay[b]	–	–	–
15	Charlotte Harbor[b]	–	–	–
16	Lake Worth Lagoon/Loxahatchee	●	●	●
17	St. Lucie Estuary	●	●	●
18	Indian River Lagoon	●	●	●
19	Halifax River		●	●
20	Guana, Tolomato, Matanzas, Pellicer System		●	●
21	St. Johns River		●	●
22	Nassau River/Big Talbot		●	●
23	St. Marys River/Amelia River		●	●

[a] Seagrass=seagrass depth; chl-a=chlorophyll *a*; DO=dissolved oxygen
[b] Estuarine systems with newly-approved state water quality standards

1.2.1. Water Clarity Targets Based on Seagrass Depth of Colonization

1.2.1.1. Background

Seagrasses are an appropriate biological endpoint for development of numeric nutrient criteria for many estuaries in Florida, because they are both ecologically important and sensitive to changes in water quality resulting from nutrient pollution. The term seagrasses refers to a class of rooted macrophytes that occur in estuaries and coastal marine waters. These species have been collectively called submerged aquatic vegetation or SAV reflecting the fact that they are not botanically classified as grasses and that freshwater species may be included as well. In this document, the term seagrasses is used, reflecting its wide acceptance and applicability to the estuarine and marine species of concern.

Seagrasses cover approximately 2.7 million acres (1990–1993 data using aerial photography) throughout the State, including portions of 18 of 23 of the estuarine systems identified by EPA (Table 1-2), and are a central ecological feature of Florida's dynamic, highly productive marine ecosystems (FFWCC 2003; Gibson et al. 2000). Seagrasses provide shelter and habitat and are productive feeding areas for many juvenile and adult aquatic species. In addition, seagrasses trap and stabilize sediments and help reduce nutrients in the water column (Gibson et al. 2000; TBEP 2000; Zieman and Zieman 1989). Healthy populations of seagrasses serve as an important and widely recognized indicator of biological integrity of estuarine systems and, in turn, of balanced natural populations of aquatic flora and fauna (Doren et al. 2009; Ferdie and Fourqurean 2004; Gibson et al. 2000; Orth et al. 2006). Historically, seagrasses have been used as an endpoint by National Estuary Programs (NEPs) and FDEP to derive recommended estuarine criteria because of the recognized importance of maintaining healthy seagrass communities in Florida.

Seagrass communities depend on a variety of physical, chemical, and biological conditions to thrive. Among these, adequate underwater light availability (as measured by water clarity) is critical for seagrass health (Dennison et al. 1993; Duarte 1991; Gallegos 2001). The relationship between water clarity and the depth to which seagrasses grow (known as the depth of colonization or Z_c) has been well-documented (Dennison 1987; Dennison et al. 1993; Gallegos 1994; Gallegos 2001; Gallegos 2005; Gallegos and Kenworthy 1996; Steward et al. 2005). These studies show that light requirements vary within a range of approximately 10 to 30 percent, with a central tendency often near 20 percent. EPA reviewed published studies of seagrass light requirements for various seagrass species and concluded that, for Florida, an average value of 20 percent of incident light penetrating from the surface to the depth of colonization is necessary for seagrasses to survive (Dennison et al. 1993; Duarte 1991; Gallegos 1994; Steward et al. 2005).

The ecological mechanisms relating estuarine water clarity to nutrient loading are well-recognized (Devlin et al. 2011; Hoyer et al. 2002). For example, increased nutrient loading may result in enhanced phytoplankton production and chlorophyll biomass. Chlorophyll is a major component of light attenuation. In addition to other water quality parameters that contribute to light attenuation (e.g., colored dissolved organic matter [CDOM] and total suspended solids [TSS]), nutrient-enhanced phytoplankton production and biomass can reduce water clarity and thus limit light availability for seagrass growth at the deep water edge.

1.2.1.2. Approach

Ecological relationships between aquatic life, seagrass habitats, and water quality provide a logical path by which water quality targets can be developed to support designated uses. A key aspect for applying the logic path is the existence of comprehensive aerial seagrass coverage maps for many estuaries in Florida, often spanning decades from the earliest maps to the most recent. The historical maps provide critical information on the potential extent and depth of seagrass beds in the absence of nutrient pollution. EPA's approach utilizes these seagrass coverage maps to develop estuary-specific targets for seagrass depth of colonization and then applies empirically-derived light requirements to compute target values for average light attenuation coefficient. Criteria for total nitrogen (TN), total phosphorus (TP) and chl-a are derived to achieve the target value for average light attenuation coefficient.

When seagrasses receive sufficient sunlight, seagrass biomass remains constant or increases over time. Conversely, when incoming light is attenuated by substances in the water column, such as phytoplankton, suspended solids, or color, seagrass growth slows or stops. The relationships of nutrient-related algal growth, reduced light availability, and reduced seagrass coverage have been broadly documented (Dennison et al. 1993; Twilley et al. 1985). Since the area within an estuary available for seagrasses to grow is, in part, a function of the total area with enough sunlight at sufficient depths to sustain growth, EPA used historical seagrass coverage to determine maximum seagrass depth of colonization in each estuary. Because seagrass habitats support a rich array of biological uses and are extremely important to the overall ecosystem health, EPA is proposing to derive numeric criteria to maintain the maximum depth of colonization to ensure protection of balanced natural populations of aquatic flora and fauna.

Depth of colonization reflects the compensation depth for seagrasses, or the maximum depth at which light is sufficient to enable production of carbon by photosynthesis to meet or exceed carbon losses. Beyond the depth of colonization, then, light is generally insufficient to maintain growth. This depth is related to the potential extent of seagrass habitats in an estuary by the amount of available habitats shallower than a given depth that have the potential to support seagrasses (e.g., Janicki and Wade 1996). In contrast, the minimum depth to which seagrasses can grow can be limited by a combination of exposure to wave energy, extreme heat or cold, ultraviolet radiation, and desiccation during low tides. When water clarity is insufficient to support seagrasses at deeper depths, the suitable habitat between shallow water and deep water limits is decreased, creating the tendency for seagrasses habitats to decrease in extent and eventually disappear (de Boer 2007; Koch 2001).

EPA calculated a segment-specific water clarity target (expressed by light attenuation coefficient) to support seagrasses at the maximum depth of colonization. This endpoint measure is derived from the water clarity needed to provide 20 percent of the incident light level at the depth of colonization for a specific segment. The water clarity target (light attenuation coefficient) was used to derive numeric criteria to support a balanced natural population of aquatic flora and fauna. Figure 1-1 shows a schematic of the methods used to derive the water clarity target (light attenuation coefficient). The details are described below.

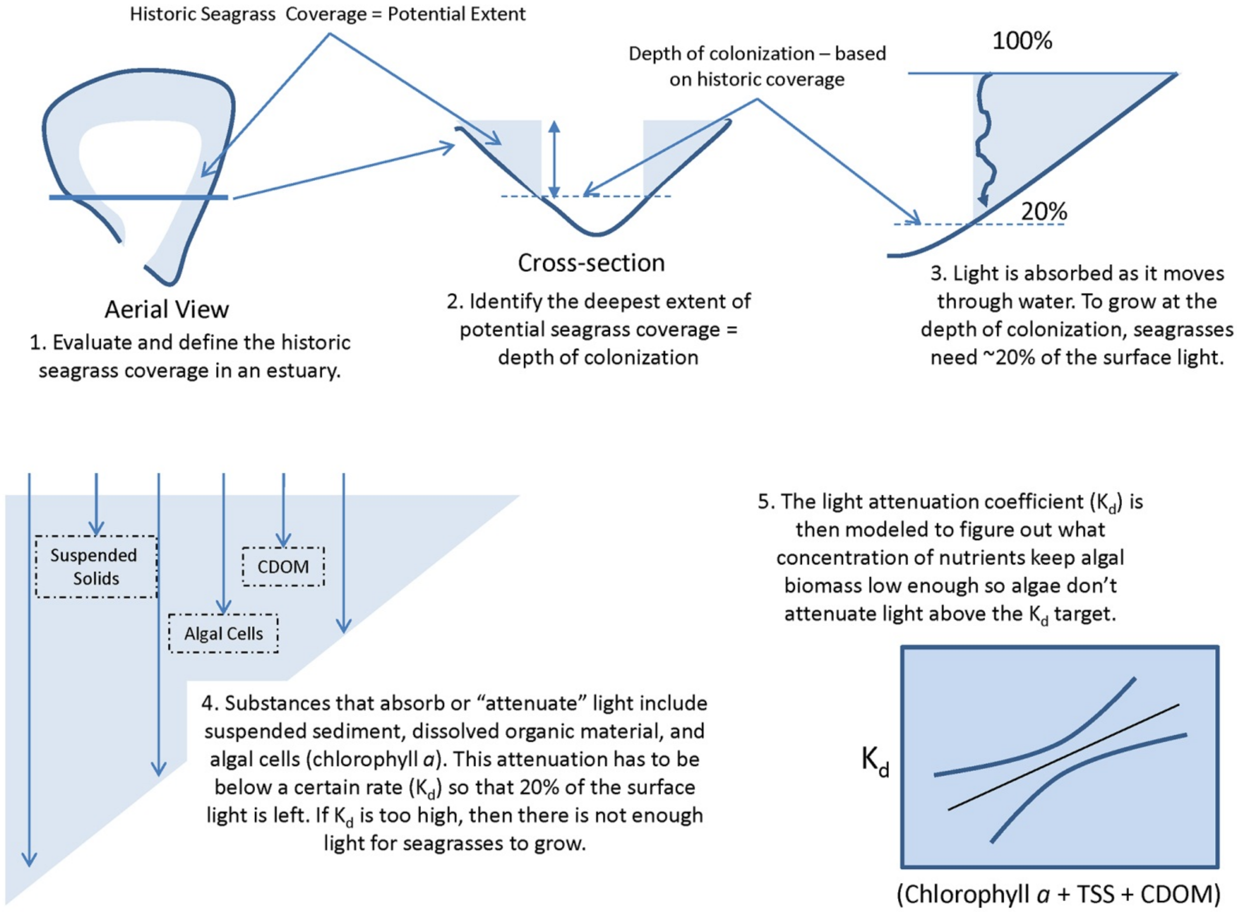


Figure 1-1. Schematic of methods used by EPA to derive the light attenuation coefficient (K_d) to reach a numeric nutrient criteria

1. Evaluate and define historic seagrass coverage in an estuary

EPA used extensive historical seagrass coverage data (the earliest available, generally 1940–1960) to compute historical depth of seagrass colonization. Seagrass coverage data were obtained from FDEP, the United States Geological Survey (USGS), the Southwest Florida Water Management District (SWFWMD), the South Florida Water Management District (SFWMD), Tampa Bay Estuary Program, Charlotte Harbor National Estuary Program, and Sarasota Bay Estuary Program. All seagrass coverages were based on interpretation of aerial imagery except for Loxahatchee Estuary, where coverage was based on field surveys. Seagrass coverage data were obtained as ESRI shapefiles delineating marine areas as un-vegetated bottom, continuous seagrass coverage, or one or more levels of patchy seagrass coverage. EPA did not distinguish between patchy seagrass coverage and continuous seagrass coverage (i.e., these were all considered seagrass) because there was not enough information available to quantify a different light requirement for supporting patchy versus continuous seagrass.

In the absence of historical data, more recent seagrass coverage data (e.g., 1992) were used. In all cases, current (2000–2010) seagrass coverage was also evaluated to determine existing depth of colonization, and to relate these values with existing water quality.

2. Identify the depth of colonization for each estuary

EPA identified the deepest extent of historical seagrass coverage as the depth of colonization endpoint measure for each estuary. EPA based the depths on one or more historical seagrass coverage maps for each estuary. In most cases, depth of colonization endpoint measure was based on the oldest, and likely least nutrient-impacted seagrass coverage. However, when the deepest depth of colonization in a segment occurred in a later year, EPA utilized the deeper value, reflecting the Agency's interpretation that this would best indicate attainment of balanced natural populations of aquatic flora and fauna in that estuary.

Seagrass depth of colonization was computed for areas of the following estuaries: Perdido Bay, Pensacola Bay, Choctawhatchee Bay, St. Andrews Bay, St. Joseph Bay, Apalachicola Bay, Apalachee Bay and the Springs Coast, Loxahatchee Estuary, and Indian River Lagoon. Seagrass coverages were based on imagery from as early as 1940 for Perdido Bay, 1943 for Indian River Lagoon, and as recently as 2010 for multiple Gulf Coast estuaries.[5]

To compute seagrass depth of colonization, EPA overlaid seagrass coverage data on bathymetric soundings compiled by the National Oceanic and Atmospheric Administration (NOAA) using a geographic information system (GIS) (Figure 1-2).[6] Bathymetric soundings within 1 km of a seagrass bed were extracted and classified as being either in seagrass or not in seagrass. Each sounding was also classified according to the estuary segment in which it is located.[7] Subsequently, soundings were binned by 25 cm depth bins and the proportion of soundings within seagrass computed by depth bin (Figure 1-3). The proportions were plotted to illustrate the decrease in seagrass cover as a function of water depth. Depth of colonization was defined as the depth at which the proportion of soundings within seagrass was reduced by 50 percent from the maximum proportion. Since the initial data were binned at 25 cm depth intervals, the correct value was computed by linear interpolation of the percentages in 25 cm bins. Validation tests demonstrated that the quantities obtained were an accurate estimate of the average depth of colonization for the segment. Local tidal information reported by NOAA[8] was used to adjust the bathymetric estimates from the original datum (such as mean lower low water or mean high water) to the mean tide level datum, which is the average water depth through which light is attenuated before reaching seagrass.

[5] Seagrass coverages used for each estuary are identified in each respective section of this document.
[6] An exception is Loxahatchee Estuary, where seagrass coverage was mapped based on a large number of direct field observations.
[7] Computations were originally performed based on FDEP's water body identification numbers (WBIDs). If the WBID coverage differed substantially from EPA's segmentation, the estimates were later re-computed for the coverage year upon which endpoint measures were based using EPA's estuary segmentation.
[8] http://www.tidesandcurrents.noaa.gov/

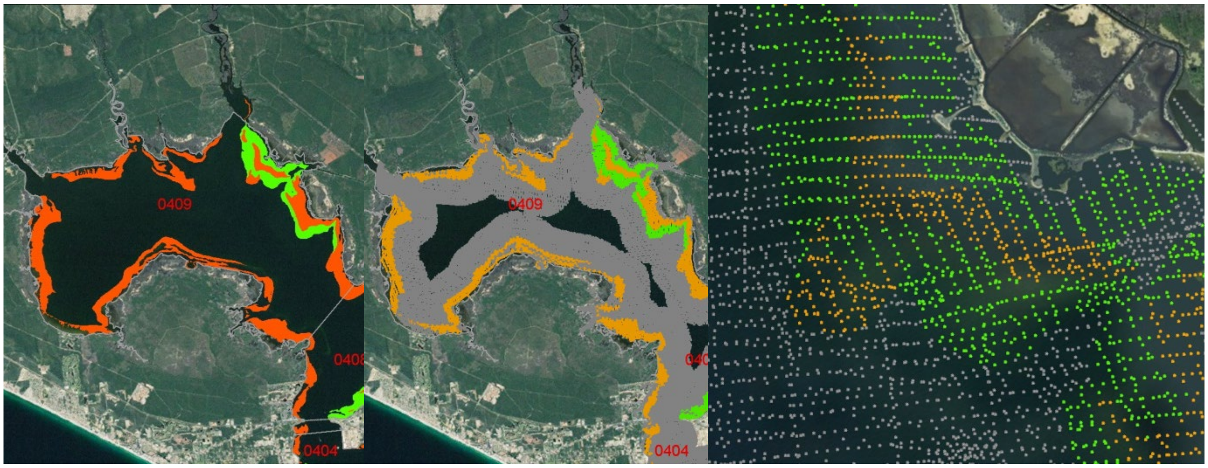


Figure 1-2. The approach for computing seagrass depth of colonization, illustrated for segment 0409 in St. Andrews Bay. (A) Seagrass coverage in the segment in 1953 (patchy seagrass = orange; continuous seagrass = green). (B) Bathymetric soundings within 1 km of the seagrass. Symbols are colored based on proximity to seagrass (green = in continuous seagrass; orange = in patchy seagrass; grey = not in seagrass. (C) A close-up from the scene in (B) illustrating the density of bathymetric soundings.

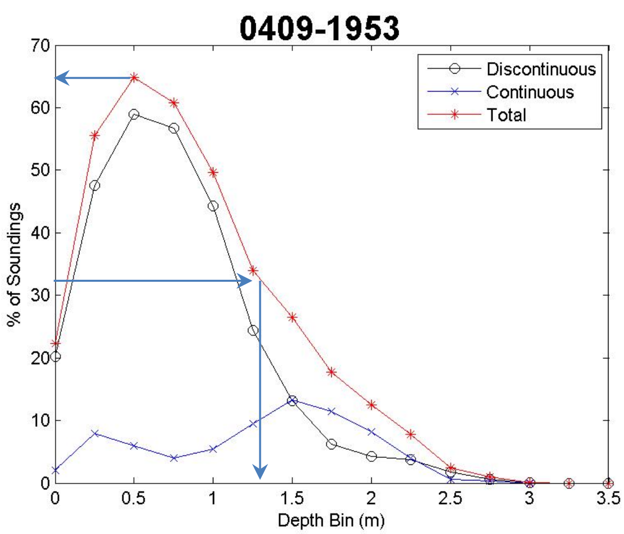


Figure 1-3. Proportion of soundings in seagrass based on the 1953 seagrass coverage, grouped by 25 cm depth bins for segment 0409 (St. Andrews Bay). The maximum proportion of soundings in seagrass was 65 percent. The depth at which this was reduced by half is 1.3 m below mean lower low water. The local difference between mean lower low water and mean tide level is 0.22 m, based on NOAA tide gauge 8729179. Therefore, the depth of colonization for this segment in 1953 was 1.52 m.

3. Light from the surface is attenuated as it passes through the water.

For seagrasses to grow at the depth of colonization, EPA determined that, for Florida, an average value of 20 percent surface light is necessary, as described above.

4. Using the requirement of 20 percent of surface light penetrating from the surface to the depth of colonization, EPA calculated a water clarity target, the average light attenuation coefficient, to support seagrasses at the depth of colonization.

Specifically, the light attenuation coefficient (K_d) necessary to achieve 20 percent of surface light (i.e., $I_{Z_c} = 0.2 \times I_0$) at the seagrass depth of colonization (Z_c) for each estuary segment was calculated using equation (1), which can be rearranged to obtain equation (2).

$$I_{Z_c}/I_0 = 0.2 = e^{-K_d Z_c} \qquad (1)$$

$$K_d = -\ln(0.2)/Z_c \qquad (2)$$

I_{Z_c} is the light intensity at the depth of colonization, Z_c, and I_0 is light intensity at the surface.

EPA validated the depth of colonization endpoint measure by evaluating light attenuation due to CDOM, denoted K_d(CDOM), with the expectation that CDOM should not attenuate light to less than 20 percent of incident light at the depth of colonization goal. K_d(CDOM) was computed from observed color data in the FDEP's Impaired Waters Rule (IWR) Run 40 database using the empirical relationship shown in equation (3). [9]

$$K_d(\text{CDOM})=0.025(\text{Color}) \qquad (3)$$

This relationship was based on a relationship between color and light absorbance at 440 nm shown below in (4) (Gallegos 2005).

$$a_g(440)=0.065\text{Color}_{diss} \qquad (4)$$

a_g is the absorbance at 440 nm due to CDOM and Color_{diss} is dissolved color.

Equation (4) is followed by conversion of $a_g(440)$ to $a_g(\text{PAR})$ using a normalized absorption function integrated across PAR wavelengths (400–700 nm) (assuming s_g=0.017, where s_g is the spectral slope) (Gallegos 2005).

5. The light attenuation coefficient target is used to compute nutrient concentrations such that algal biomass does not reduce water clarity to a point of degrading seagrass habitats. See Section 1.4.1.3 for analytical approaches used to derive nutrient criteria.

[9] Where possible, average color and salinity were computed by estuary segment. The regression relationship between average salinity and average color was used to predict average color by segment. Because salinity has been sampled more regularly than color, this approach provides a better estimate of segment average K_d(CDOM) than the color data alone.

1.2.2. Maintenance of Balanced Algal Populations

1.2.2.1. Background

A principal route through which nutrient pollution affects designated uses is the increase in chl-a concentrations resulting from enhanced algal growth and biomass accumulation in estuarine surface waters. The use of chl-a as an algal biomass indicator is a scientifically defensible metric well-studied in estuarine ecology (Boyer et al. 2009; Cullen 1982; Day et al. 1989; Steele 1962;). EPA has previously identified balanced algal populations as an appropriate endpoint for the development of numeric nutrient criteria in estuarine waters (USEPA 2000a, 2000b). Furthermore, the state of Florida currently utilizes a chl-a concentration to assess estuarine water quality (Section 62-303, F.A.C.).

In estuaries, the nutrient-driven effects on algal growth and biomass accumulation can result in more frequent, short term blooms of one or more algal species. These events can decrease water clarity and adversely affect aesthetics, recreation, and aquatic life habitat. They can also be manifested as harmful algae, which can produce toxins that adversely affect both human health and aquatic life. Frequent algal blooms can also result in longer term increases in average chl-a concentration and thus affect the longer term balance of organic matter cycling within an estuary (Nixon 1995). Excess organic matter loading can often lead to hypoxia or anoxia, which also can adversely affect habitat and aquatic life.

The temporal scale over which excess algal biomass affects designated uses varies by use and causal path. For example, for swimmers or those fishing the average annual chl-a concentration is likely less important than conditions that exist on the day they wish to fish or swim (Walker 1985). On the other hand, changes in water clarity over extended periods are likely more detrimental to long-term impact on aquatic life through habitat disturbance than any single excursion. That is because it is a longer-term decline in light availability that will ultimately decrease growth and survival of aquatic flora, like seagrasses. Numeric nutrient criteria for estuaries were derived to protect all Class II and III uses, which include recreation, propagation and maintenance of a healthy, well-balanced population of fish and wildlife. Deriving chlorophyll criteria on the basis of reducing the likelihood of algal blooms is a viable approach for deriving criteria given the effect of nuisance algal blooms on recreation and recreational uses (Larkin and Adams 2007; Walker 1985) as a reflection of floral and faunal balance.

Based on scientific literature, the current application of chl-a as a water quality management tool, and observed chl-a concentrations in Florida estuaries, EPA concluded that the prevention of frequent algal blooms in Florida estuaries is important in supporting balanced populations of aquatic flora and fauna, and in turn protecting Florida's Class II and Class III designated uses in Florida estuaries.

1.2.2.2. Approach

The concept of deriving protective chl-a concentrations and associated nutrient targets on the basis of reducing nuisance chl-a levels has been applied to lakes (Walker 1985), including some lakes in Florida (Bachmann et al. 2003; Havens and Walker 2002; Walker and Havens 1995). Specific chl-a concentrations consistent with nuisance conditions were defined in that literature on the basis of trophic state boundaries, user perception studies, and observed impacts.

For estuaries, the literature is limited on trophic state chl-a thresholds. NOAA used its' Assessment of Estuarine Trophic Status to determine that estuarine chl-a concentrations during the annual bloom period could be used as an indicator of algal bloom conditions. Low algal bloom conditions were defined as maximum chl-a concentrations < 5 µg/L, medium bloom conditions as maximum chl-a concentrations 5–20 µg/L, high bloom conditions as maximum chl-a concentrations 20–60 µg/L, and hypereutrophic conditions as maximum chl-a concentrations above 60 µg/L (Bricker et al. 2003). The United Kingdom Comprehensive Studies Task Team took the original freshwater trophic state categories (OECD 1982) and extended them to estuaries using a maximum summer chl-a value of 10 µg/L as an estuarine eutrophic threshold (Painting et al. 2007); that was extended in a later study to a maximum summer chl-a value of 10 µg/L for offshore marine water bodies and 15 µg/L for nearshore marine water bodies (Painting et al. 2005; Tett et al. 2007).

Elevated chl-a concentrations in lakes and reservoirs are often associated with the dominance of nuisance algal species including cyanobacteria such as *Microcystis, Anabaena,* and *Aphanizomenon* (Chorus et al. 2000). Some of these species are known to produce toxins, which are harmful to humans and aquatic life. Studies have shown that increased chl-a concentrations manifested as nuisance algal blooms may also occur within estuaries, embayments, and nearshore waters (Anderson et al. 2008; Bricker et al. 2007; Paerl et al. 2008; Steidinger et al. 1999;). Ongoing research continues to explore the relationships among nutrient pollution, increased primary production, and harmful algae in marine systems across the United States (U.S.) and worldwide. EPA evaluated the existing scientific information and determined that frequently occurring elevated chl-a concentrations can be an expression of dominance by one or more phytoplankton species, potentially toxic or otherwise harmful or nuisance algae, and these conditions likely represent an imbalance in the natural populations of aquatic life in Florida estuaries.

To set a chl-a concentration target, EPA utilized information on bloom frequencies typical of Florida estuaries and then identified concentrations typical of blooms of harmful or nuisance algae and indicative of imbalance of phytoplankton populations. One estimate for the range of observed monthly chl-a maxima was from 15 to 25 µg/L, depending on the type of estuary (coastal embayment, river-dominated, or lagoon) (Glibert et al. 2010). In a national survey, the average bloom chl-a concentrations were 20 µg/L or less for 7 of 10 large estuaries; concentrations were especially low for Florida Bay (8 µg/L) and Pensacola Bay (10 µg/L, Glibert et al. 2010) and higher for the St. Johns River Estuary (20 µg/L, Bricker et al. 2007).

EPA selected a chl-a concentration target of 20 µg/L as that which defines nuisance conditions for use in deriving numeric criteria for estuaries based on the scientific literature. EPA also defined an allowable probability of exceedance for the chlorophyll concentration target by considering the acceptable risk and existing literature and selected a value of no more than 10 percent. A probability of exceedance of less than 10 percent is consistent with target chlorophyll levels observed for lake systems being managed for nutrient pollution in Florida (Havens and Walker 2002). Higher averages were deemed to be inconsistent with other management goals and with minimizing risk to regular recreation and aquatic life use.

1.2.3. DO Targets

1.2.3.1. Background

An additional endpoint used by EPA in setting nutrient criteria for Florida estuaries was DO concentration. Adequate DO is necessary to protect aquatic life and balanced community composition, and the effects of DO on water quality for CWA purposes have been well studied (USEPA 1976, 1986a, 1986b, 2000c). Florida currently has DO criteria for protection of its designated uses (Subsection 62-302.530(30), F.A.C.).

DO concentrations can be significantly impacted by high anthropogenic nutrient loadings to coastal waters. The relationship between nitrogen and phosphorus pollution and marine hypoxia is clear and well documented in the scientific literature (Conley et al. 2009a, 2009b; Diaz 2001; Diaz and Rosenberg 2008). Excess nutrients increase phytoplankton blooms, and this increase in organic material drives microbial decomposition and consumes DO (Dodds 2006). Oxygen is depleted when primary production, stimulated by nutrients, exceeds consumption. Biochemical oxygen demand (BOD) increases particularly in the bottom layer of the water column because density stratification leads to limited mixing and reaeration. Cases of bottom layer hypoxia or anoxia caused by eutrophication are becoming more common in estuaries and coastal systems around the world (Breitburg et al. 2003; Diaz and Rosenberg 2008). Hypoxia is typically defined as DO < 2 mg/L, and anoxia as DO < 0.1 mg/L (USEPA 1999).

Increases in algal growth and subsequent reductions in DO impact many aspects of estuary and coastal communities. In estuaries and coastal waters, low DO is one of the most widely reported consequences of nitrogen and phosphorus pollution and one of the best predictors of a range of biotic impairments (Bricker et al. 2003). Low DO concentrations reduce the extent and quality of habitat for a variety of organisms (Baden et al. 1990; Breitburg 2002; Diaz and Rosenberg 2008; Rabalais et al. 2001). Low DO causes impacts to aquatic life ranging from mortality to chronic impairment of growth and reproduction (USEPA 2001).When nutrient pollution creates conditions that result in large hypoxic zones, substantial changes in fish, benthic, and plankton communities may occur (Howell and Simpson 1994; Kidwell et al. 2009). This includes avoidance of these zones by fish, mobile benthic invertebrates migrating from the hypoxic area, and fish kills in some systems when fish and other mobile aquatic organisms have nowhere to migrate away from the areas with low DO (Howell and Simpson 1994; Kidwell et al. 2009). This can result in changes to the benthic invertebrate community structure of estuaries and coastal areas, with increases of organisms more tolerant of low DO (Baker and Mann 1992, 1994a, 1994b; Baustian and Rabalais 2009; Breitburg 2002). Even intermittent hypoxia can cause shifts in the benthic assemblage to favor resistant or tolerant organisms, which are less desirable food sources, creating unbalanced benthic communities in the hypoxic zone because fish avoid the area (Kidwell et al. 2009). When hypoxia or anoxia extends into shallow waters, it affects spawning and nursery areas for many important fish species by reducing the habitat available that protects smaller fish and aquatic organisms, especially juveniles, from predation (Breitburg 2002). Reduced fishery production in hypoxic zones has been documented in the U.S. and worldwide (Diaz and Rosenberg 2008).

Hypoxia and anoxia in bottom waters also result in anoxia in surface sediments, which has geochemical consequences including acidification, and the release of toxic hydrogen sulfide, soluble reactive phosphorus (SRP), and ammonia (NH_3) (Cai et al. 2011; Diaz and Rosenberg 2008; Kemp et al. 2005; McCarthy et al. 2008;). The sediment of hypoxic zones then becomes a potential source of nutrients that can increase the degree of eutrophication. As a result, systems that have had persistent and chronic hypoxia often fail to recover quickly even after nutrient pollution loadings have been reduced (Conley et al. 2007). Reduced oxygen also affects a variety of other biogeochemical processes that can impact water quality, such as the chemical form of metals in the water column (Snoeyink and Jenkins 1980).

1.2.3.2. Approach

EPA selected a DO metric that represents support of healthy aquatic floral and faunal communities as a third biological endpoint to derive numeric criteria for estuaries. As described in more detail above, DO concentrations are a well-known indicator of the health of estuarine and coastal biological communities. Aquatic animals including fish, benthic macroinvertebrates and zooplankton depend on levels of DO that meet the needs of all species and life stages (e.g., larval, juvenile, and adult) (Diaz 2001; Diaz and Rosenberg 2008).

To maintain aquatic flora and fauna, EPA conducted an analysis of the DO requirements of sensitive species using the Virginian Province DO evaluation procedure (USEPA 2000c; Vincent et al. in review). This procedure was developed as a recommended approach for deriving the lower limits of DO necessary to protect coastal and estuarine animals in the Virginian Province (Cape Cod, MA, to Cape Hatteras, NC). However, with appropriate modification, the procedure may be applied to other coastal regions of the United States. The approach to determine the limits of DO that will protect saltwater animals within the Virginian Province considers both continuous (i.e., persistent) and cyclic (e.g., diel) exposures to low DO. Both scenarios cover three areas of protection including juvenile and adult survival, growth effects, and larval recruitment effects (USEPA 2000c).

The results of this analysis were used to determine whether DO targets considered for numeric nutrient criteria development would ensure the protection of sensitive aquatic life in Florida estuaries. Based on levels of DO that meet the needs of sensitive aquatic animal species in Florida estuarine waters, EPA selected a minimum allowable water column average DO of 4.0 mg/L, a minimum daily water column average DO of 5.0 mg/L, and a minimum bottom water average DO of 1.5 mg/L in estuarine waters for use in deriving numeric criteria. These values and interpretations are also consistent with existing Florida DO criteria (Subsection 62-302.530(30), F.A.C.) and FDEP's assessment process (Subsection 62-303.320(5), F.A.C.).

Empirical sample data and modeling data were also treated consistently with the provisions provided in Subsection 62-303.320(4), F.A.C., namely:

> (4)(a) Samples collected at the same location less than four days apart shall be considered as one sample, with the median value used to represent the sampling period. However, if any of the individual DO values are less than 1.5 mg/L or, for other parameters, individual values exceed acutely toxic levels […], then the worst case value shall be used to represent the sampling period. The worst case value is the minimum value for DO, both the minimum and maximum for pH, or

the maximum value for other parameters. However, when DO data are available from diel or depth profile studies, the lower tenth percentile value shall be used to represent worst case conditions for comparison against the minimum criteria.

(b) Samples collected within 200 m of each other will be considered the same station or location, unless there is a tributary, an outfall, or significant change in the hydrography of the water.

(c) Samples collected from different stations within a water segment shall be assessed as separate samples even if collected at the same time.

(d) In making the determination to list water segments, the Department shall consider ambient background conditions, including seasonal and other natural variations.

1.3. Classification and Segmentation

1.3.1. Classification

EPA is proposing to develop numeric nutrient criteria for Florida's estuaries on a system-specific basis. A system-specific approach allows the Agency to consider the individual characteristics of the integrated watershed-estuarine ecosystems, their specific water quality conditions, aquatic life attributes, and the estuary-specific responses to nutrient inputs. Delineating Florida's estuarine waters in this manner provides EPA with an organizational framework for developing and presenting the scientific approach, applying the methods and approaches appropriate to each estuary, and ultimately deriving numeric criteria. EPA's proposed classification (i.e., estuary delineation) approach is based on the natural geographic attributes of estuarine basins and their associated watersheds. Natural barriers between estuarine basins tend to limit water flow and exchange between estuaries, even if exchanges are not eliminated entirely. This general classification approach has been used previously in developing the NOAA Coastal Assessment Framework (Bricker et al. 1999). Using this approach, EPA classified 23 estuarine areas in Florida (Table 1-3)—including 8 in the Florida Panhandle region, 3 in the Big Bend region, 4 in southwest Florida, and 8 on the Atlantic Coast (see Figure 1-4).[10] As noted in the introduction to Volume 1, Section 1, Florida has numeric nutrient criteria for four of the water bodies covered by this classification. As a result, EPA is proposing numeric criteria for 19 of the 23 estuarine systems.

[10] A total of 26 estuarine and coastal areas are identified. EPA's approach for offshore coastal waters is presented in Volume 2 of this Technical Support Document (TSD).

Table 1-3. Identification of estuarine systems in Florida

Estuary Number	Estuary Names
1	Perdido Bay
2	Pensacola Bay
3	Choctawhatchee Bay
4	St. Andrews Bay
5	St. Joseph Bay
6	Apalachicola Bay
7	Alligator Harbor
8	Ochlockonee Bay
9	Big Bend
10	Suwannee Sound
11	Springs Coast
12	Clearwater Harbor/St. Joseph Sound[a]
13	Tampa Bay[a]
14	Sarasota Bay[a]
15	Charlotte Harbor[a]
16	Lake Worth Lagoon/Loxahatchee
17	St. Lucie Estuary
18	Indian River Lagoon
19	Halifax River
20	Guana, Tolomato, Matanzas, Pellicer System
21	St. Johns River
22	Nassau River/Big Talbot
23	St. Marys River/Amelia River

[a] Estuarine systems with newly-approved state water quality standards

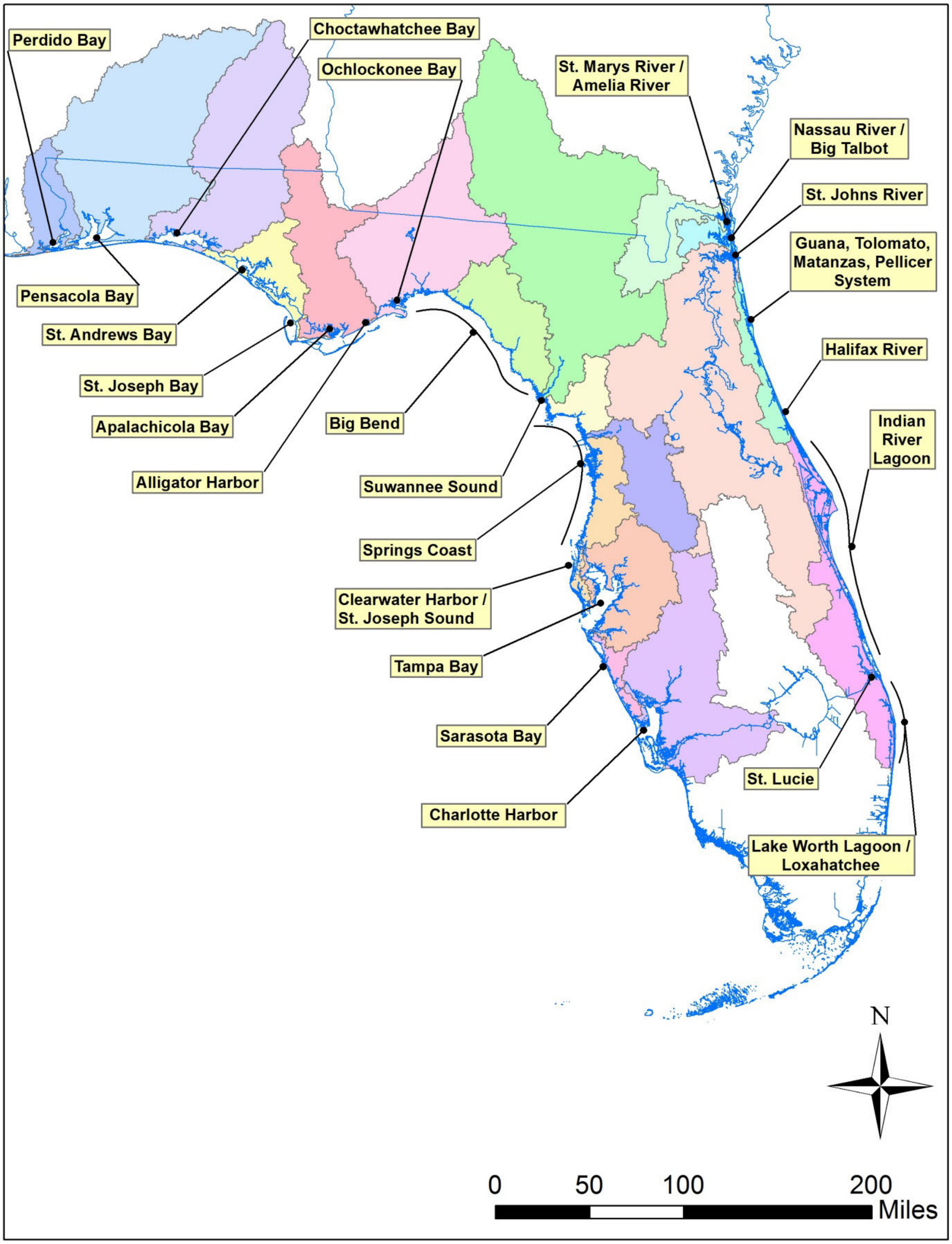


Figure 1-4. Delineation of estuarine waters in Florida and their associated watershed boundaries by color; the watershed for south Florida estuaries is shown in white

1.3.2. Segmentation

The concentrations of water quality constituents are variable in estuaries because of the interactions of several factors such as tides, freshwater flows, wind, and many other physical, chemical, and biological attributes. Recognizing that water quality targets and criteria designed to meet the targets may vary along the estuarine gradient, each estuary was subdivided into segments for analysis. The approach is similar to that used by Florida's NEPs (e.g., Tampa Bay, Sarasota Bay, Charlotte Harbor, and the Indian River Lagoon) for water quality monitoring, reporting, and management purposes.

In developing a uniform approach that could be applied to the 19 estuarine systems identified, patterns in long-term average salinity distributions were used as the primary basis for segmentation. Salinity reflects the effects of physical processes mixing fresh and salt water and by-and-large characterizes the predominant ecological zones within the estuary. In other words, salinity is a conservative measure of the physical exchanges between marine and freshwater sources of nutrients. Typically, salinity in an estuary increases with distance from a freshwater source such as a river. Thus, an estuary can be segmented on the basis of major salinity gradients from the head of the estuary to open sea. Salinity in the estuary is also affected by episodic climate events, such as El Niño, hurricanes, and high rainfall events, that impact freshwater discharge and mixing within the estuary. Long-term average salinity distributions were used to reduce the effects of such episodic anomalies. Contours of average salinity or isohalines were used to demarcate zones of average salinity gradients in the estuary.

Average salinity distributions were derived from IWR Run 40 data (FDEP 2010). Station-specific average salinity was imported into the mapping program ArcGIS along with monitoring station location. A continuous raster surface of salinity was interpolated using the Inverse Distance Weighted algorithm in the Spatial Analyst tool in ArcGIS. The Inverse Distance Weighted algorithm calculates the value at points between stations using surrounding values, and continuous salinity contour lines or isohalines are drawn and mapped using the interpolated values.

In addition to salinity, other factors were also considered when segmenting each estuary to account for hydrology and ecosystem dynamics. First, physical features such as bridges and causeways were used to aid in delineation where they notably affected the hydrodynamic circulation in the estuary. Second, seagrass coverage and depth distribution was used because seagrass is a fixed biological community that reflects specific salinity zones. Because biological communities reflect the conditions present in an area over time they can be useful tools to help distinguish regions of the estuary. Last, EPA consulted with the NEPs (e.g., Tampa Bay, Indian River Lagoon, Charlotte Harbor, and Sarasota Bay) on the approaches used to derive their existing segmentation schemes in those systems.

In most cases, segment boundaries were nearly parallel to the isohalines. Segment shape files were created in ArcGIS for each estuary to provide a consistent process for analyzing the individual segments. A total of 89 segments in 19 estuaries were used as the basis for EPA's analysis (see Table 1-4). Portions of the segments that were not classified as marine waters (Class III marine) were removed in ArcGIS. EPA gave each segment a unique, 4-digit segment number in which the first two digits represent the estuary, and the last two digits represent the segment.

Table 1-4. Segments derived for each estuary system

Estuary Number	Estuary Names	Segments
1	Perdido Bay	4
2	Pensacola Bay	9
3	Choctawhatchee Bay	3
4	St. Andrews Bay	9
5	St. Joseph Bay	1
6	Apalachicola Bay	5
7	Alligator Harbor	3
8	Ochlockonee Bay	5
9	Big Bend	8
10	Suwannee Sound	1
11	Springs Coast	14
12	Clearwater Harbor/St. Joseph Sound[a]	–
13	Tampa Bay[a]	–
14	Sarasota Bay[a]	–
15	Charlotte Harbor[a]	–
16	Lake Worth Lagoon/Loxahatchee	6
17	St. Lucie Estuary	3
18	Indian River Lagoon	6
19	Halifax River	2
20	Guana, Tolomato, Matanzas, Pellicer System	2
21	St. Johns River	3
22	Nassau River/Big Talbot	3
23	St. Marys River/Amelia River	2

[a] Estuarine systems with newly-approved state water quality standards

1.4. Analytical Approaches Used to Derive Numeric Nutrient Criteria for Florida Estuaries

EPA re-evaluated the methods and approaches presented to and evaluated by the Science Advisory Board (SAB) (USEPA 2010a) to derive numeric nutrient criteria for estuaries that included reference condition approaches, statistical approaches (i.e., stressor-response) and approaches using water quality simulation models. Given the variation in historical data and known nutrient-related impairments among estuaries, use of the reference condition approach was not pursued to derive numeric nutrient criteria for Florida estuaries because reference conditions that reflected minimally impacted or least disturbed conditions could not be identified. Statistical and water quality simulation models were used separately or in combination to derive the proposed numeric nutrient criteria on the basis of the estuary-specific data available for analysis. This section describes the two methods EPA applied for deriving numeric nutrient criteria for Florida estuaries.

Although water quality models are fundamentally different from statistical models, the conceptual approach that EPA proposes for both approaches is very similar. Specifically, biological endpoints and associated water quality targets are used to determine protective concentrations of TN, TP, and chl-a to derive numeric criteria for the purposes of the CWA. The statistical and mechanistic models are used to determine the TN and TP concentrations necessary to achieve water quality targets for the selected biological endpoints, and derive numeric criteria, including the magnitude, frequency, and duration components of water quality criteria.

EPA proposes that for a given estuary, the TN, TP, and chl-a content must not exceed the applicable criterion concentration more than once in a 3-year period. EPA is proposing criteria duration of a year, in which sampled nutrient concentrations are summarized as annual geometric means, because annual average concentrations are directly related to annual nutrient loading to the water body. EPA has determined that such a frequency of exceedances would not result in unacceptable effects on designated uses because it would allow estuaries enough time to recover from occasionally elevated levels of nitrogen and phosphorus concentrations.

Table 1-5 provides a summary of the number of segments, endpoints/targets, and analytical approaches used in each estuary.

Table 1-5. Summary of estuary system segmentation, endpoints, and analytical methods

			Endpoints/Targets[a]			Analytical Methods	
Estuary System		**Segments**	**Seagrass**	**Chl-a**	**DO**	**Mechanistic**	**Empirical**
1	Perdido Bay	4	●	●	●	●	●
2	Pensacola Bay	9	●	●	●	●	●
3	Choctawhatchee Bay	3	●	●	●	●	●
4	St. Andrews Bay	9	●	●	●	●	●
5	St. Joseph Bay	1	●	●	●	●	●
6	Apalachicola Bay	5	●	●	●	●	●
7	Alligator Harbor	3	●	●	●	●	●
8	Ochlockonee Bay	11	●	●	●	●	●
9	Big Bend	2	●	●	●	●	●
10	Suwannee Sound	1	●	●	●	●	●
11	Springs Coast	14	●	●	●	●	●
12	Clearwater Harbor/St. Joseph Sound[b]	–	–	–	–	–	–
13	Tampa Bay[b]	–	–	–	–	–	–
14	Sarasota Bay[b]	–	–	–	–	–	–
15	Charlotte Harbor[b]	–	–	–	–	–	–
16	Lake Worth Lagoon/Loxahatchee	6	●	●	●	●	●
17	St. Lucie Estuary	3	●	●	●	●	●
18	Indian River Lagoon	6	●	●	●		●
19	Halifax River	2		●	●		●
20	Guana, Tolomato, Matanzas, Pellicer System	2		●	●		●
21	St. Johns River	3		●	●	●	●
22	Nassau River/Big Talbot	3		●	●	●	●
23	St. Marys River/Amelia River	2		●	●	●	●

[a] Seagrass=seagrass depth; chl-a=chlorophyll *a*; DO=dissolved oxygen
[b] Estuarine systems with newly-approved state water quality standards

1.4.1. Water Quality Simulation Modeling

1.4.1.1. Models Used

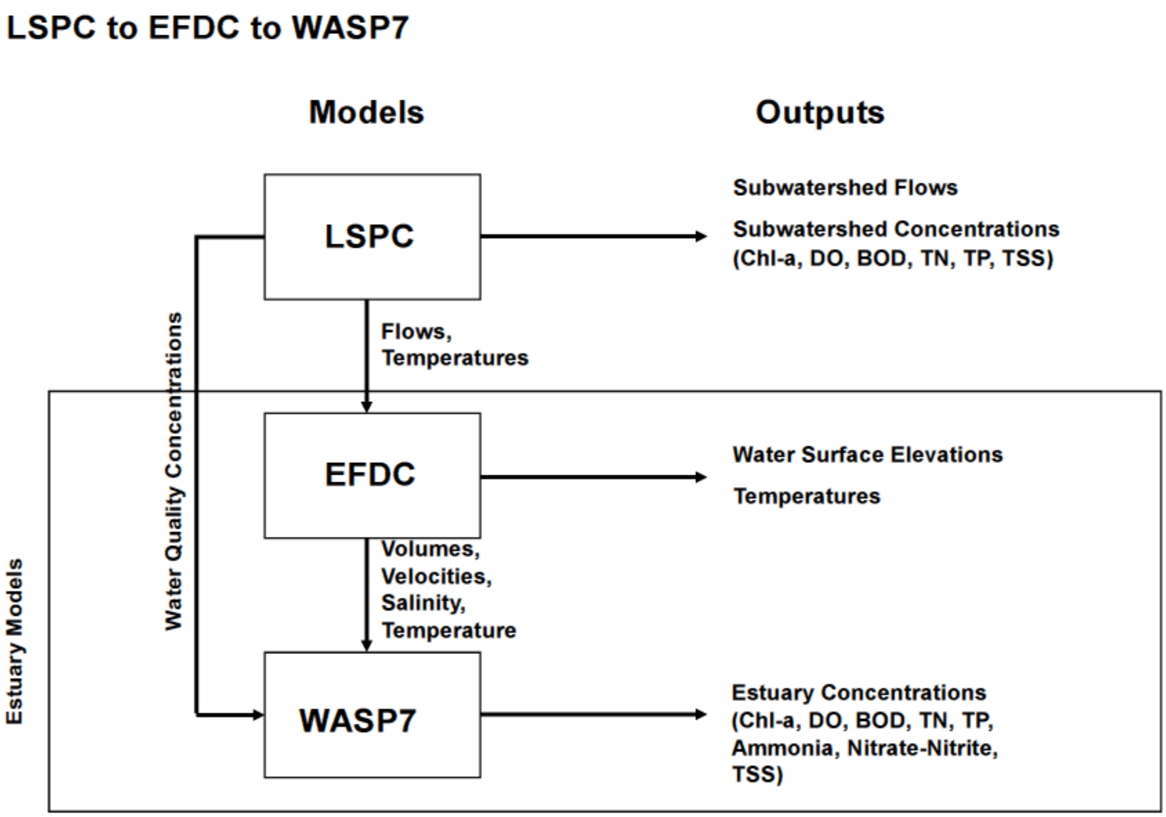
Water quality simulation models are mathematical expressions of water movement and water quality processes. They are used to simulate the relationships between nutrient inputs and the water quality responses. EPA used watershed simulation models integrated with coupled estuarine hydrodynamic and water quality simulation models to simulate the physical, chemical, and biological processes influencing watershed nutrient delivery and nutrient-related responses in Florida estuaries. Because, at present, no single computer model simulates watershed and estuary processes, three computer models were linked and used as the mechanistic model relating nutrient levels in the estuaries to the water quality targets.

Similar approaches to the one proposed by EPA have been utilized in Florida, the most well-known of which may be the hydrodynamic water quality model application in the St. Johns River and its watershed (Tillman et al. 2004; FDEP 2008).

To consider which models could be used to develop numeric criteria, EPA developed an inventory of the watershed and estuary models that had been previously applied to estuaries in Florida. EPA's inventory was based on a review of models developed by FDEP or used in Florida (Wolfe 2007), with additions based on discussions with FDEP staff (for a review of the modeling tools that were considered, see USEPA 2010a, Appendix C). On the basis of the review, EPA selected the following models to simulate stream hydrology and water quality in watersheds, as well as hydrodynamics and water quality in estuaries:

- The Loading Simulation Program in C++ (LSPC)
- The Environmental Fluid Dynamics Code (EFDC)
- The Water Quality Analysis Simulation Program Version 7.3 (WASP7)

Figure 1-5 shows how the three models interact with one another. LSPC simulates the hydrological and water quality conditions in the watersheds. EFDC uses flow rates and temperatures output by LSPC and tidal stage and salinity data as boundary conditions to calculate velocities, temperature, and salinity within cells (volume elements) in the estuary at each simulation time step. Inputs to WASP7 include stream discharges of freshwater, nutrients, chl-a, DO, oxygen demand, and suspended solids from LSPC and velocities and temperatures from EFDC. WASP7 outputs include nutrients, chl-a, and suspended solids concentrations in each grid cell for each time step, along with numerous additional water quality data.



BOD=biochemical oxygen demand; chl-a=chlorophyll *a*; DO=dissolved oxygen; TN=total nitrogen; TP=total phosphorus; TSS=total suspended solids

Figure 1-5. Linkage between LSPC, EFDC, and WASP7 models

Sixteen LSPC watershed models were developed, representing the 16 major watersheds in Florida's panhandle, northern, and central regions, with the exception of the Lake Okeechobee watershed (Figure 1-6), as described in Appendix C: Watershed Hydrology and Water Quality Modeling Report for Florida Watersheds. The outputs from the LSPC watershed models were input to the EFDC and WASP7 estuary models. For the nine estuary systems, nine EFDC models and nine WASP7 models were developed (Table 1-6). Appendix D: Hydrodynamic and Water Quality Modeling Report for Nutrient Criteria for Florida Estuary Systems includes both the hydrodynamic and water quality calibration and validation results for these nine estuary systems.

A similar suite of interconnected, basinwide hydrologic, hydrodynamic, and water quality models were previously developed to model the Lower St. Johns River for other water quality management purposes. Given the similarities of the pre-existing watershed and estuary models to those being developed for others estuarine systems, the Pollutant Load Simulation Model (PLSM) watershed model, and EFDC/ICE- estuary models were used to derive numeric nutrient criteria for St. Johns River estuary (FDEP 2008, see Appendix G).

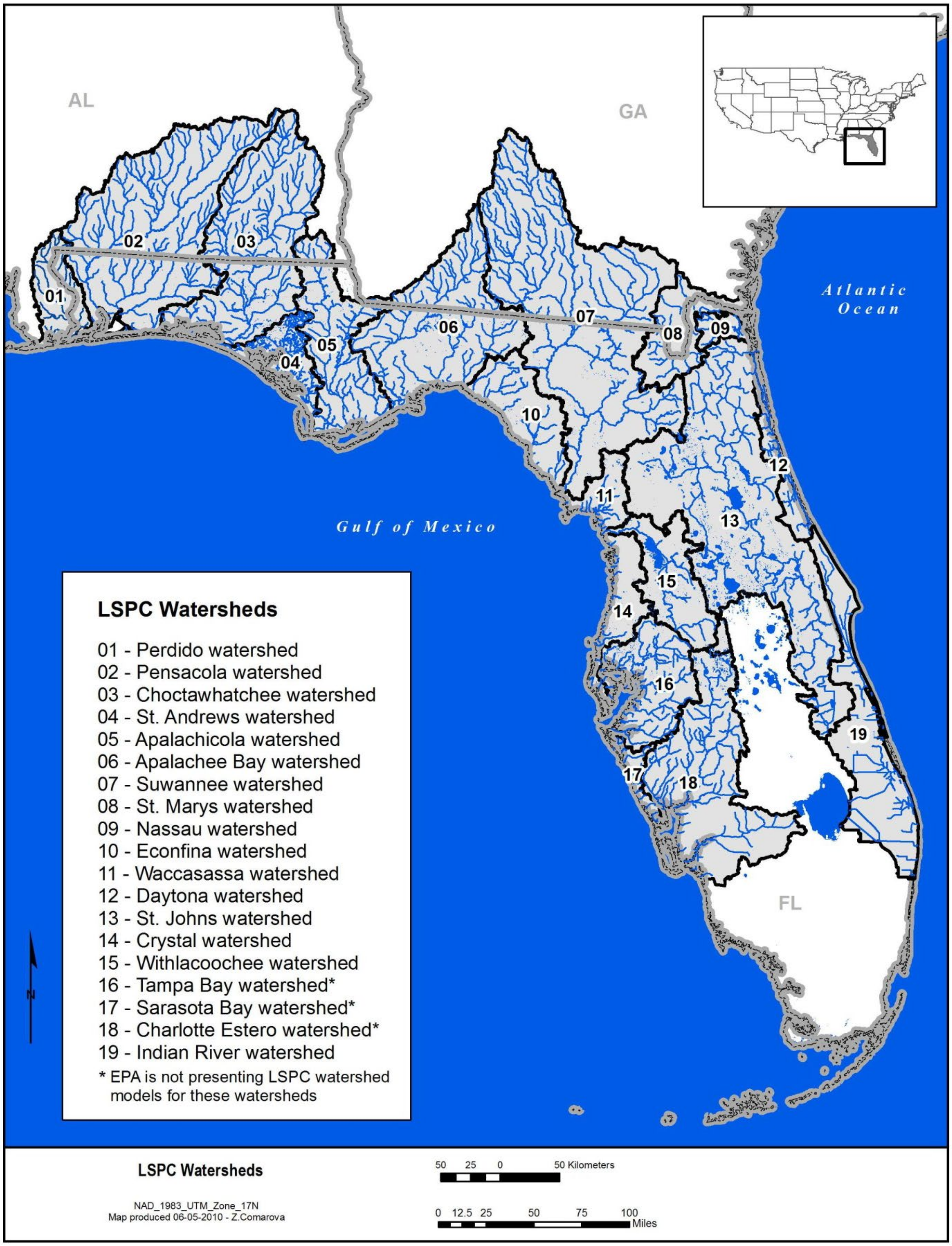


Figure 1-6. Location of Florida watersheds and their watershed numbers

Table 1-6. Relationship between estuary systems, EFDC and WASP7 estuary models, and LSPC watershed models

Estuary System		EFDC and WASP7 Estuary Models	LSPC Watershed Models
1	Perdido Bay	Perdido Bay	Perdido
2	Pensacola Bay	Pensacola Bay	Pensacola
3	Choctawhatchee Bay	Choctawhatchee Bay	Choctawhatchee
4 5	St. Andrews Bay St. Joseph Bay	St. Andrews Bay	St. Andrews
6 7 8 9 10 11 12	Apalachicola Bay Alligator Harbor Ochlockonee Bay Big Bend Suwannee Sound Springs Coast Clearwater Harbor/St. Joseph Sound[a]	Big Bend	Apalachicola Apalachee Suwannee Econfina Waccasassa Crystal Withlacoochee
13 14	Tampa Bay[a] Sarasota Bay[a]	–	–
15	Charlotte Harbor[a]	–	–
16[b]	Lake Worth Lagoon/Loxahatchee	(1) Lake Worth Lagoon (2) Loxahatchee River	Indian River
17	St. Lucie Estuary	St. Lucie	Indian River
18	Indian River Lagoon	Not modeled	Indian River
19	Halifax River	Not modeled	Daytona
20	Guana, Tolomato, Matanzas, Pellicer System	Not modeled	Daytona
21	St. Johns River	EPA's TMDL[c]	St. Johns
22 23	Nassau River/Big Talbot St. Marys River/Amelia River	Nassau and St. Marys	Nassau St. Marys

[a] Estuarine systems with newly-approved state water quality standards
[b] Separate EFDC and WASP7 estuary models were developed for Lake Worth Lagoon and Loxahatchee River
[c] TMDL=total maximum daily load

1.4.1.2. Model Calibration and Validation[11]

Watershed (LSPC) and estuary hydrodynamics and water quality models (EFDC/WASP7) were calibrated and validated prior to use in deriving numeric nutrient criteria. Calibration entails adjustment of model parameters to match model outputs to observed values. Validation entails application of the calibrated model to inputs different from those used for calibration and comparison of output quantities with observed values. These processes ensure the mechanistic models faithfully simulate the physical, chemical and biological processes in the watersheds and estuaries, provide justification for extending the models to conditions outside those for which the model is calibrated, and allow objective evaluation of the model's performance. Calibration and validation are best practices in mechanistic modeling and examples of these procedures may be found in prior studies using EFDC (e.g., Liu et al. 2008; Xia et al. 2011; Yang et al. 2007; Zou et al. 2006).

[11] Model validation, also known as confirmation testing, uses independent data to determine whether the model is predictively valid and to identify the range of conditions under which cause and effect relationships can be determined (McCutcheon et al. 1990).

Calibration of the LSPC watershed hydrology model involved adjustment of the model parameters (such as infiltration and groundwater recession rates) such that the simulated stream flows adequately matched the streamflows measured at USGS flow stations. The calibration of the hydrologic parameters was performed using the available data from January 1, 1997 through December 31, 2009. All USGS flow gages were analyzed to determine which gages were useful for hydrology calibration based on three criteria: (1) gage data available for the entire simulation period, (2) gage data not tidally influenced, and (3) gage location not downstream of control structures. If a gage station did not meet all three criteria, the calibration of the hydrology model was not focused on that particular station. However, the stations not used as calibration stations were still used in the validation process. During the modeling process, if the selected gages did not adequately represent the varied topographical features, such as land uses or soil groups, additional stations were selected and used as validation stations. To support the calibration of the watershed model, validation stations were used to help confirm the calibration. A rating system was applied to the calibration and validations stations to determine the overall calibration success. A weighted score was assigned to simulated versus observed errors, with total flow, storm flow, and low-flow volumes having the greatest weight. Consistent with standard modeling practices, the summation of the weighted scores was assigned a qualitative descriptor of Very Good (VG), Good (G), Fair (F), or Poor (P). The highest possible score was 80, and the lowest possible score was 20. Scores of 80–76 were rated as VG, 75–56 as G, 55–36 as F, and 35–20 as P. This scoring system has been applied in other modeling projects in the southeastern United States (e.g., Floyds Fork, Kentucky; Carter Lake, Coosa River, and Chattahoochee River watersheds, Georgia) (Tetra Tech, Inc. 2011).

The calibration of the LSPC water quality model involved adjustment of the model parameters (such as build-up rates and groundwater concentrations) such that simulated water quality concentrations and loads (specifically for TP, TN, and BOD) adequately matched measured water quality concentrations and loads. Both visual inspection and statistical metrics were used during calibration. Visual calibration was accomplished by matching the trends in the measured water quality concentration data to observed water quality data based on a visual comparison of the two. Loading metrics, including annual loading percent error, were used for statistical calibration. A rating system was applied to the percent error of the average annual loadings at the calibration and validation stations to determine the overall calibration success. The average annual loading percent error was assigned a qualitative descriptor of Very Good (VG), Good (G), Fair (F), or Poor (P). Scores of 0–40 percent were rated as VG, 40–90 percent as G, 90–150 percent as F, and 150–500 percent as P, using procedures set forth in McCutcheon et al. (1990).

Calibration of the estuary model using available data in the period January 1, 2002 to December 31, 2005, was a hierarchical process, beginning with calibration of water surface elevation (WSE) in EFDC; WSE is the major forcing factor of water dynamics in Florida estuaries. In this first step, the open boundary tidal forcing was adjusted in amplitude and phase such that predicted hourly tidal station stage measurements matched observed values. The next step of involved calibration of EFDC salinity and water temperature dynamics. Both visual and statistical measures were used in this and subsequent estuary model calibration steps. Validation of the EFDC model was conducted using comparisons of simulated data with independent data (different monitoring stations' locations or periods of observation).

The calibrated and validated EFDC model produced velocities, water temperatures, salinities, and volumes for use in the water quality model (WASP7) calibration and validation. WASP7 was calibrated by adjusting values of numerous biological and chemical parameters used in the mathematical equations describing chemical and biological transformations of the model constituents. Success of calibration was measured by comparing predicted DO, chl-a, TN, ammonia, nitrate+nitrite (NO_3+NO_2), TP, TSS, and light extinction coefficient to observed values.

Table 1-7 and Table 1-8 present the rating system for determining the overall success of calibration-validation for the hydrodynamics and water quality models of Florida estuaries. The rating system was based on EPA technical guidance for model applications (Donigian 2000; McCutcheon et al. 1990). The rating categories represent qualitative descriptors based on the percent mean differences between simulated and observed values.

Table 1-7. Calibration/validation ratings for EFDC/WASP7 applications for Florida estuaries

	Percent Difference between Simulated and Observed Values		
State Variable	**Very Good**	**Good**	**Fair**
Salinity	< 15	15–25	25–40
Water temperature	< 7	7–12	12–18
Water quality/DO	< 15	15–25	25–35
Nutrients/chl-a	< 30	30–45	45–60

Table 1-8. Relative errors and statistical targets for hydrologic calibration (Lumb et al. 1994)

Relative Errors (Simulated-Observed)	Statistical Target (%)
Error in total volume	10
Error in 50% lowest flows	10
Error in 10% highest flows	15
Seasonal volume error - Summer	30
Seasonal volume error - Fall	30
Seasonal volume error - Winter	30
Seasonal volume error - Spring	30
Error in storm volumes	20
Error in summer storm volumes	50

1.4.1.3. Water Quality Simulation Modeling with Endpoints that Demonstrate Support of the Estuarine Designated Use

The process that EPA is proposing for developing numeric criteria using water quality simulation models involves simulating the watershed loading to the estuary with hydrologic watershed models and modeling the estuarine water quality response with the hydrodynamic and water quality models. Different nutrient loading conditions were evaluated to determine the levels that result in support of designated uses. The calibrated models were used to evaluate various nutrient loading conditions and the corresponding estuary response. The predicted estuary conditions were compared to three endpoint targets. The endpoints consist of a light attenuation metric that represents maintenance of healthy seagrass communities, a DO metric that represents maintenance of healthy aquatic life, and a phytoplankton biomass/algal bloom metric that represents maintenance of balanced algal populations.

Average light attenuation, which is measure of water clarity, has been shown to be a good predictor of seagrass depth of colonization. As described in Section 1.2.1.2 above, scientific literature indicates that seagrasses require, on average, about 20 percent of incident surface light at the depth of colonization. Historic seagrass depths were determined from aerial photos of seagrass beds and depth measurements. Light attenuation coefficients (K_d) necessary to achieve 20 percent of surface light at the target seagrass depth of colonization (Z_c) were calculated as $K_d = -\ln(0.2)/Z_c$. In the water quality simulation modeling approach, watershed nutrient loads were decreased in the model to decrease light attenuation and allow more light through the water column until the targeted light attenuation coefficient was met.

The second endpoint target used to derive numeric nutrient criteria is based on phytoplankton biomass and preventing excessive algal blooms. As described in Section 1.2.2.2 of this document, scientific literature supports the use of chl-a concentrations of 20 µg/L. EPA is applying a chl-a endpoint that prevents concentrations above 20 µg/L more than 10 percent of the time. Specifically, the modeled 90th percentile average daily concentration in the surface layer must be less than or equal to 20 µg/L to be considered supporting designated uses.

In addition to the light attenuation coefficient/seagrass depth of colonization target and the chl-a target, the water quality simulations were used to model the DO endpoint target. In developing that endpoint target, EPA considered both the draft manuscript *Dissolved Oxygen Requirements of Florida-Resident Saltwater Species Applied to Water Quality Criteria Development* by Vincent et al. (in review), and the existing Florida DO criteria (Subsection 62-302.530(30), F.A.C)]. Vincent et al. (in review) describes an analysis in which the authors computed acute DO criteria of 2.79 mg/L for juveniles and adults and 3.41 mg/L for larvae, as well as a criteria value of 5.0 mg/L to protect against chronic DO effects. Separate criteria were identified for Atlantic and Short-nose Sturgeon, sensitive aquatic species present in many Florida estuaries. These criteria, include a value of 3.2 mg/L applicable when water temperatures were not stressful (22–26 °C) and 4.2 mg/L applicable to warmer, more stressful water temperatures (Campbell and Goodman 2004). In the analysis, sensitivity test data were assembled and reviewed to identify Florida-relevant data, and the Virginian Province approach was applied. Florida DO criteria include an allowable minimum DO of 4.0 mg/L and a daily average of 5.0 mg/L. A provision is provided in Subsection 62-303.320(4) F.A.C., which states that individual DO values should not be below 1.5 mg/L, and personal communication with FDEP staff indicated that a criterion value of 1.5 mg/L is also applied to waters that experience vertical DO stratification. The minimum

DO of 4.0 mg/L from FDEP's criteria was applied as an acute DO endpoint, the value of 5.0 was used as a chronic DO endpoint, and 1.5 mg/L was also used as an endpoint target to protect against hypoxic events. In summary, for this numeric nutrient criteria development process, EPA is applying a 10th percentile water column average DO of 4.0 mg/L, a 10th percentile daily average of 5.0 mg/L, and a lower vertical layer average of 1.5 mg/L. Each of those three DO endpoint targets is applied as estuary segment averages.

In the water quality simulation models, the light, chl-a, and DO endpoint targets were applied in accordance with available data and species present. Model simulations began with applying the calibrated watershed, hydrodynamic, and water quality models to the 2002–2009 period for each estuary system. Next, water quality models were used to simulate estuarine responses in the absence of anthropogenic nutrient inputs. These background or non-anthropogenic nutrient conditions were achieved in the model by setting all point source nutrient inputs to zero and by setting all anthropogenic land uses (e.g., urban lands, agricultural lands, etc.) to forest or wetland land use, thereby reducing nonpoint source nutrients inputs. The watershed TN and TP loading rates were used in the hydrodynamic and water quality models to simulate the water quality response and biological endpoint conditions that would be expected in the estuary under current and background conditions. EPA used the models to determine the levels of TN and TP necessary to meet the biological endpoints and support the designated uses.

Model sensitivity tests were performed to evaluate the extent to which the chl-a, light attenuation, and DO endpoint measures respond to changes in nutrients. Sensitivity tests are discussed more in the following section. Based on model sensitivity results, model simulations deemed to meet the endpoints in each estuary segment were used to calculate numeric nutrient criteria. In some estuarine segments, model simulations were unable to achieve one or more endpoints within the constraints of the nutrient inputs established in the model simulations. That is, some segments did not meet the endpoints at the non-anthropogenic nutrient levels. In those cases, EPA found that factors other than nutrients played a significant role in light, chl-a, or DO dynamics; factors such as CDOM, suspended sediment, and low DO from watershed processes would need to be controlled to meet the endpoint targets. Therefore, for such cases, endpoint targets that could be met with nutrient reductions became the primary endpoint targets and provided the primary line of evidence. EPA used model simulation results that achieved the most stringent endpoint targets to calculate annual geometric means for each of the years 2002 through 2009. Candidate numeric nutrient criteria for TN, TP, and chl-a were then calculated as the 90th percentile of these geometric means for each estuary segment. EPA selected the 90th percentile to characterize the upper bound of conditions supporting designated uses.

1.4.1.4. Sensitivity Analysis

Sensitivity analysis is a useful tool for calibrating a model and understanding the estuary response to different causative parameters. In a sensitivity analysis, model parameters and input data are varied individually to determine which parameter or boundary condition causes the greatest change in the model simulation. EPA used sensitivity analysis to determine the change in estuary water quality related to a change in the input nutrient loading. In this sensitivity test, 2002–2009 model nutrient levels were decreased by 30 percent and increased by 30 percent and the simulated change in the endpoint measures was noted. The three endpoint measures are average water column light attenuation coefficient, 90th percentile of surface layer chl-a concentration and 10th percentile of DO concentration. EPA calculated the normalized sensitivity coefficient (S) as the ratio of fraction of change in the endpoint measure to the fraction of change in the nutrient loads according to the following equation which is similar to the equation incorporated into the EPA QUAL2E water quality model (Brown and Barnwell 1987):

$$S = \frac{\frac{\Delta y}{y}}{\frac{1}{2}\left\{\frac{\Delta TN}{TN} + \frac{\Delta TP}{TP}\right\}}$$

Where Δy is the change in the endpoint measure value (e.g., chl-a concentration), y is the baseline value for the endpoint measure, ΔTN is the change in TN load and ΔTP is the change in TP load.

The results of model sensitivity analyses indicated that in general chl-a was highly sensitive to nutrient inputs in most estuary segments (Figure 1-7), light attenuation was moderately sensitive (Figure 1-8), and DO was only slightly sensitive in two estuaries (Pensacola Bay and Big Bend estuaries, Figure 1-9).

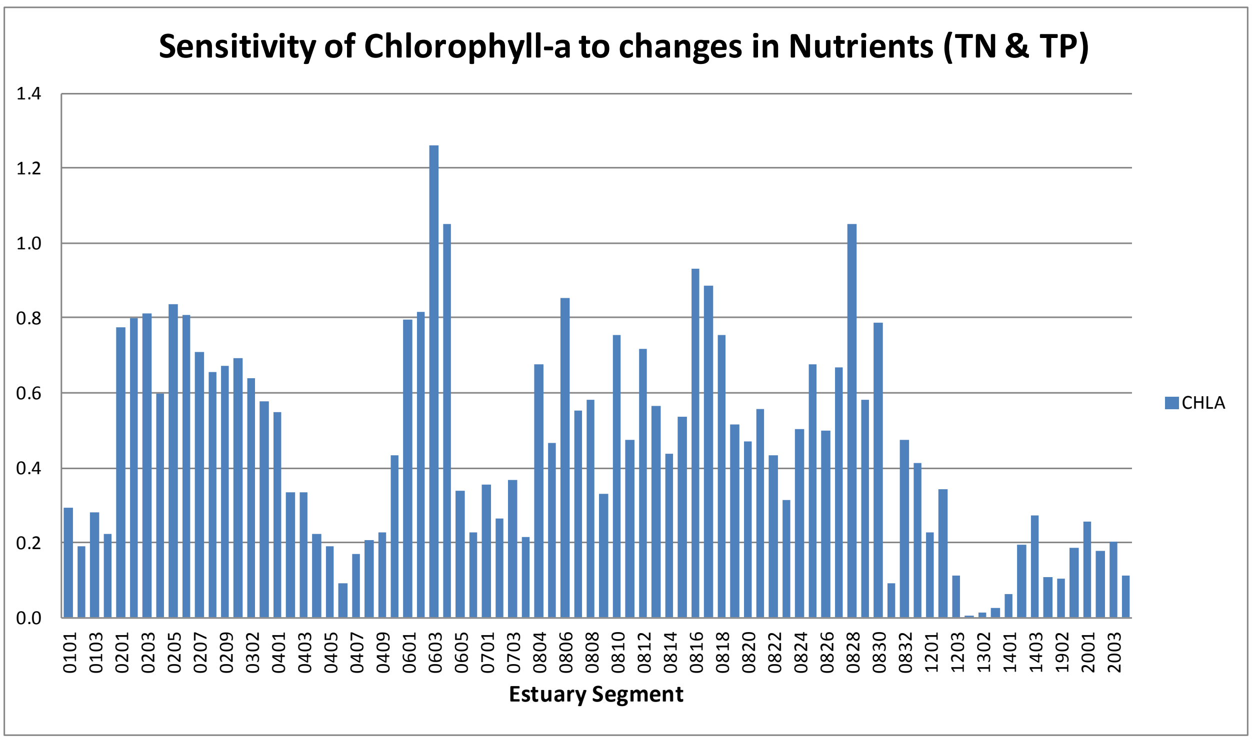


Figure 1-7. Estuary model sensitivity of chl-a to nutrient changes

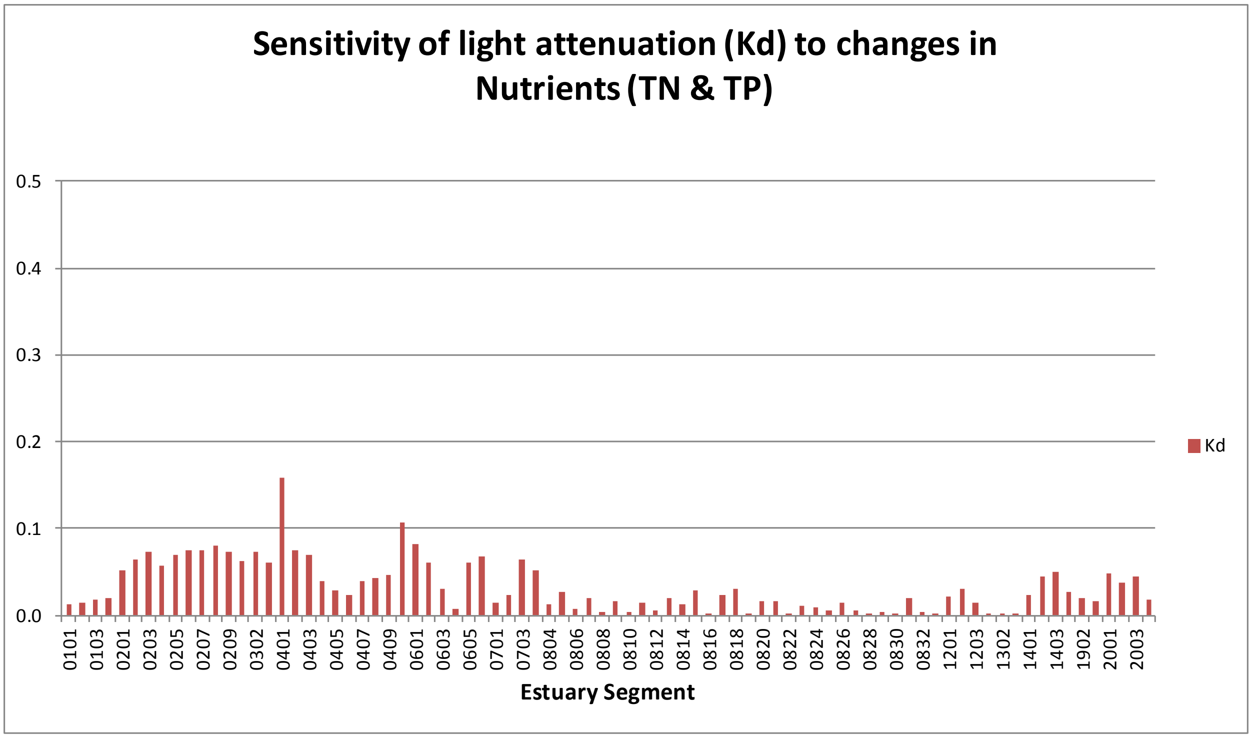


Figure 1-8. Estuary model sensitivity of light attenuation coefficient (K_d) to nutrient changes

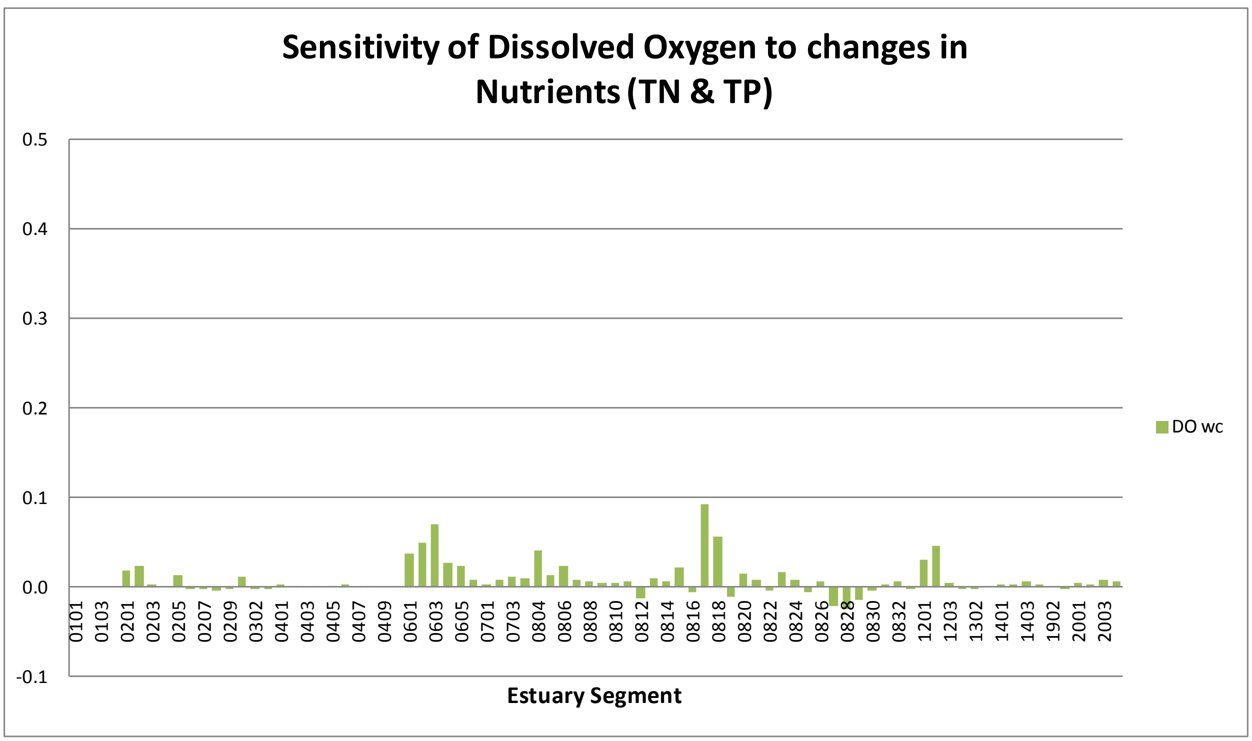


Figure 1-9. Estuary model sensitivity of water column DO to nutrient changes

The sensitivity of chl-a to nutrients is explained by algal growth kinetics and the Michaelis-Menton formulation used in the water quality model. In this model formulation, the nutrient effect on algal growth is greatest at low levels of nutrients and then tapers off asymptotically as the nutrient levels increase above the Michaelis constant (also called the half saturation constant). Light and temperature also affect the algal growth rate, so any lack of sensitivity could be due to concentrations of nutrients being much greater than the Michaelis constant, or temperature or light effects.

The sensitivity of light attenuation to nutrients is as expected for most of the estuaries. Light is affected by CDOM, solids, as well as phytoplankton chl-a. Nutrients affect light indirectly through their effect on phytoplankton growth. Because of the effect of these other constituents on light and the indirect nature of nutrient effects, the sensitivity of light to nutrients is much less than the sensitivity of phytoplankton chl-a to nutrients.

The nutrient effect on DO is also multifaceted. Nitrogen directly affects DO through nitrification and indirectly through phytoplankton production, respiration, death and decay. In addition to these nutrient effects, DO is affected by reaeration, carbonaceous biochemical oxygen demand (CBOD), and sediment oxygen demand (SOD). Because of the influence of these other factors on DO, the sensitivity of DO to nutrient loading may be limited.

The information from the sensitivity analysis was used by EPA to guide the use and interpretation of model results. For example, the model was not used to evaluate nutrient reduction scenarios for endpoints found to be insensitive to nutrient changes. In cases where endpoints were sensitive to nutrient changes, EPA used the model to evaluate nutrient reductions. In addition to evaluating the general sensitivity of endpoints to changes in nutrients, EPA also limited the water quality model analysis by only evaluating nutrient levels above non-anthropogenic levels. If the endpoint showed slight sensitivity to changes in nutrients, but not enough for the non-anthropogenic nutrient levels to result in an endpoint meeting the target, then EPA did not rely on that endpoint as a primary line of evidence.

1.4.2. Statistical Models (Stressor-Response)

Analyses of field collected measurements yield relationships between nutrient concentrations and endpoint targets (i.e., stressor-response relationships, USEPA 2010b) which are useful for developing numeric nutrient criteria.

Detailed descriptions of the data and statistical methods used to calculate numeric nutrient criteria for each estuary are provided in Appendix B: Statistical (Stressor-Response) Analysis. Here, only a brief overview of each of the statistical analyses is included.

1.4.2.1. Data

Water quality data were retrieved from IWR Run 40 (FDEP 2010) and screened based on whether they were located within pre-defined estuary segments, yielding a total of 1,126,560 water quality samples collected at 26,495 sites. Different screens (described below) were applied to these data depending on the requirements of different analyses, which reduced the size of the database.

Measurements of TN, TP, chl-a, nitrate+nitrite, and total Kjeldhal nitrogen (TKN) were all log-transformed to reduce the skewness of the distributions.

Measurements of Secchi depth were converted to an estimate of the light attenuation coefficient (K_d) using the following relationship (Holmes 1970):

$$Kd = \frac{1.44}{Secchi\ depth}$$

1.4.2.2. Data Screening and Sufficiency for Analysis

Data available for each estuary segment were evaluated in terms of temporal and spatial representativeness. In general, at least two spatially distinct sampling locations, each with at least 3 years of matched TN, TP, and chl-a data were required for each segment. Relationships between TN, TP, and chl-a concentrations were estimated statistically in estuaries in which all segments met these minimal data sufficiency requirements. Similarly, to statistically estimate relationships between light attenuation coefficients and chl-a, at least 1 year of matched Secchi depth and chl-a data at two distinct sampling locations was required for each segment. Fewer years of data at each sampling location were required to estimate the light attenuation coefficient–chl-a relationship because it was anticipated that differences in this relationship among stations would be much smaller than differences in TN–TP–chl-a relationships.

1.4.2.3. Derivation of Chl-a Criteria to Meet Water Clarity Targets Based on Seagrass Depth of Colonization

Chl-a criterion values that achieve the water clarity targets were derived by first using a linear mixed model to estimate a relationship between annual average light attenuation coefficient and annual average values of chl-a, turbidity, and color. The linear mixed model accounted for hierarchical structure of the data within stations and within segments, allowing relationships between light attenuation coefficient, chl-a, turbidity, and color to vary among stations and among estuary segments, while still ensuring that these different relationships were related. Candidate chl-a criteria were then calculated as the chl-a concentrations associated with the point at which mean annual average light attenuation coefficient was equivalent to that required to achieve 20 percent light at the seagrass depth of colonization target, assuming that turbidity and color were equal to respective long-term average values.

In some cases, candidate chl-a criteria derived from the relationship between light attenuation coefficient and chl-a were outside the range of geometric mean values observed in the data set. Since the reliability of the statistical model to make accurate predictions decreases substantially outside the range of available data, candidate criteria in these cases are based instead on the limits of the data. More specifically, if a candidate chl-a criterion derived from the light attenuation coefficient–chl-a relationship was greater than upper bound of the available data, then the candidate criterion was set at the upper bound. Similarly, if a candidate chl-a criterion derived from the light attenuation coefficient–chl-a relationship was less than the lower bound of the available data, then the candidate criterion was set at the lower bound of the available data (Figure 1-10).

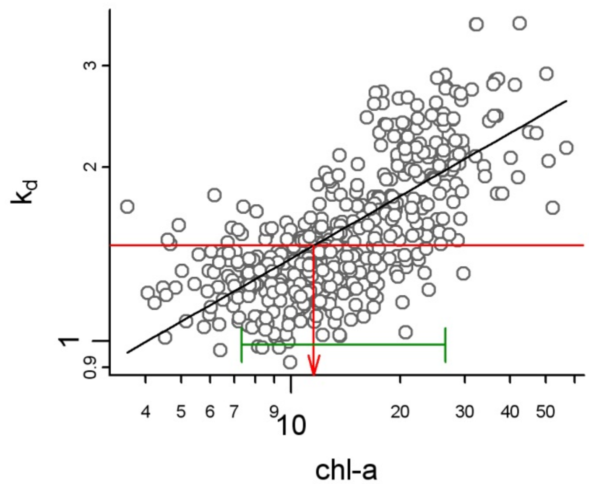


Figure 1-10. Example of estimated relationship between chl-a and light attenuation coefficient (K_d). Red horizontal line shows light attenuation coefficient corresponding with seagrass depth of colonization target for the segment. Red vertical arrow show annual geometric mean chl-a concentration predicted to be associated with light attenuation coefficient target. Green line segment shows the 5th to 95th percentile range of observed annual geometric mean values for chl-a. Open circles show measured annual average values of light attenuation coefficient (adjusted for the effect of turbidity and color) and chl-a.

1.4.2.4. Derivation of Chl-a Criteria to Meet Water Quality Targets to Maintain Balanced Algal Populations

Linear mixed models were also used to estimate the relationship within which chl-a concentrations exceeded 20 µg/L (as described in Section 1.2.2) as a function of annual geometric mean chl-a, TN, and TP concentrations. As with the statistical models described in the previous sections, the linear mixed model takes into account the hierarchical structure of the data within stations and segments. To derive candidate criteria associated with maintaining balanced algal populations, the fitted model was used to calculate annual geometric mean concentrations of chl-a, TN, and TP associated with a bloom probability of 10 percent.

In some cases, candidate chl-a criteria derived from the relationship between algal bloom frequency and chl-a were outside the range of geometric mean values observed in the data set. Since the reliability of the statistical model to make accurate predictions decreases substantially outside the range of available data, candidate criteria in these cases are based instead on the limits of the data. More specifically, if a candidate chl-a criterion derived from the bloom frequency–chl-a relationship was greater than upper bound of the available data, then the candidate criterion was set at the upper bound. Similarly, if a candidate chl-a criterion derived from the bloom frequency–chl-a relationship was less than the lower bound of the available data, then the candidate criterion was set at the lower bound of the available data (Figure 1-11).

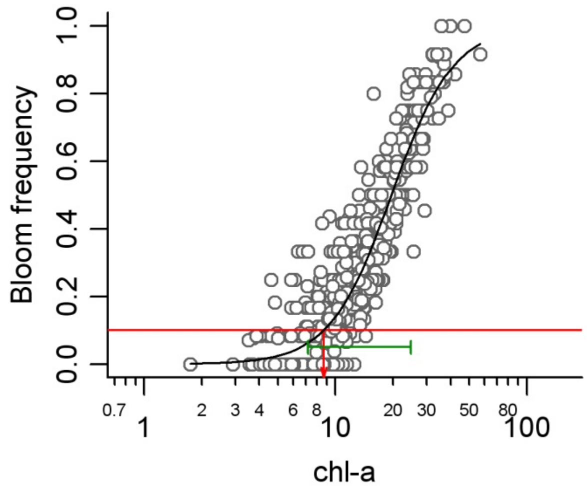


Figure 1-11. Example of model of the relationship between annual geometric mean chl-a concentration and bloom frequency. Open circles: observed annual frequency of chl-a concentrations exceeding 20 µg/L versus annual geometric mean chl-a concentration for the same year; solid black line: modeled relationship between mean chl-a concentration and bloom frequency; horizontal red line: targeted bloom frequency of 10%; vertical red arrow: annual geometry mean chl-a concentration associated with targeted bloom frequency; green line segment: 5th to 95th percentile range of observed annual geometric mean chl-a concentrations.

Relationships between light attenuation coefficient and chl-a varied among estuaries and estuary segments (Figure 1-12).

In segments in which the slope of the relationship between light attenuation coefficient and chl-a was very near zero (such that the confidence intervals included zero) or negative, water clarity was noted as being insensitive to changes in chl-a, and no chl-a criterion value associated with the water clarity endpoint was computed.

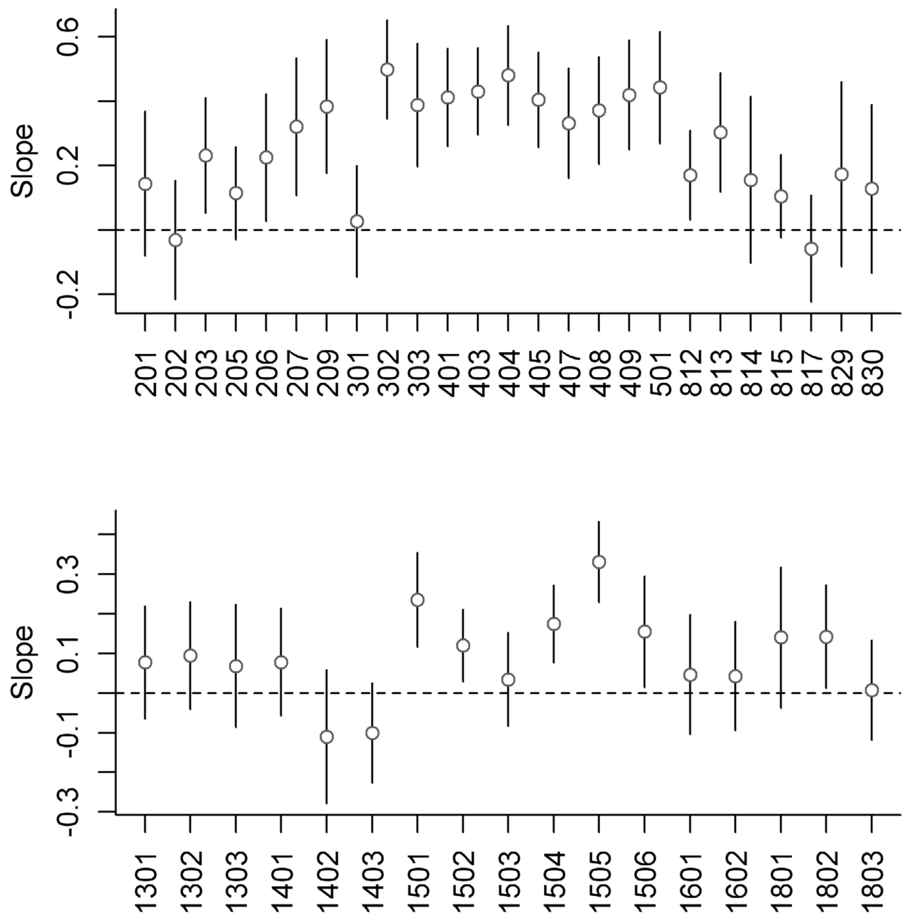


Figure 1-12. Estimates of the slopes between light attenuation coefficient and chl-a in different estuarine segments. Two panels from top to bottom show Northwest and Eastern estuaries. Vertical lines show estimates of the 90% confidence intervals on each slope, open circles show mean estimate of slope.

1.4.2.5. Derivation of TN and TP Criteria to Meet Chl-a Criteria

After chl-a criteria were derived to either meet the water quality target based on seagrass depth of colonization or to maintain balanced algal populations, nutrient–chl-a relationships were used to derive TN and TP concentrations necessary to reduce chl-a to its criterion value. A mixed model was used to estimate relationships between nutrient concentrations and chl-a concentrations, allowing the hierarchical nature of the data within stations and within estuary segments to be taken into account. This model estimated the linear relationship between annual geometric mean chl-a and annual geometric mean TN and TP at different stations. Annual geometric mean values at each station were first computed from individual sampled values. Because relationships developed for stations within the same estuary segment are likely to be similar, the hierarchical model assumes that model coefficients estimated from each station were drawn from a single, common normal distribution. This partial pooling allowed estimates of relationships at stations with less data to borrow information from stations with relatively more data (Gelman and Hill 2007).

As with the use of other statistically estimated stressor-response relationships, in some cases the candidate TN or TP criteria computed from the stressor-response relationship may be outside the range of available data. In these cases, the criterion is set at the upper or lower bound of the available data (Figure 1-13).

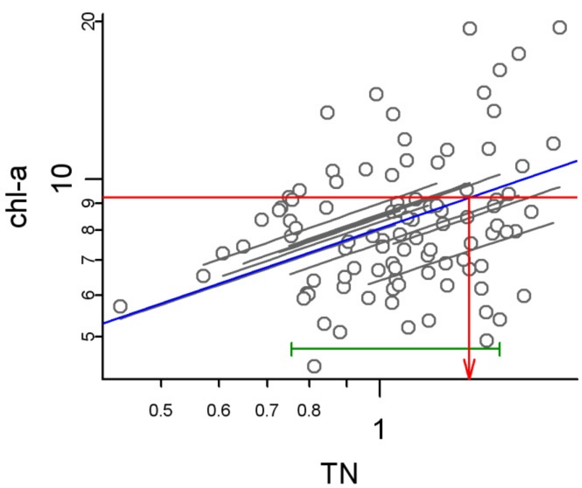


Figure 1-13. Example of estimated stressor-response relationship between TN and chl-a. Blue line: segment mean relationship. Grey lines: relationships estimated within different stations. Red horizontal line shows the chl-a criterion associated with the water clarity endpoint for each segment. Green line segment shows the 5th to 95th percentile range of observed annual geometric mean TN values. Open circles show observed values of annual geometric mean chl-a and TN.

1.4.2.6. Interpreting the Results of Statistical Analysis

Estimates of the slope of the relationship between TP and chl-a were strongly positive for the vast majority of estuary segments (Figure 1-14). The only segments in which near zero or negative slopes were observed were 1801 and 1802. In contrast, a number of segments exhibited near zero (such that confidence intervals included zero) or negative slopes of the relationship between TN and chl-a (Figure 1-15). Negative correlations between TN and chl-a likely reflect the complexities of the components of TN that are available for biological uptake. In certain locations, it seems likely that the majority of measured TN is composed of biologically unavailable forms. Further analysis of relationships between nitrate+nitrite and chl-a support this interpretation.

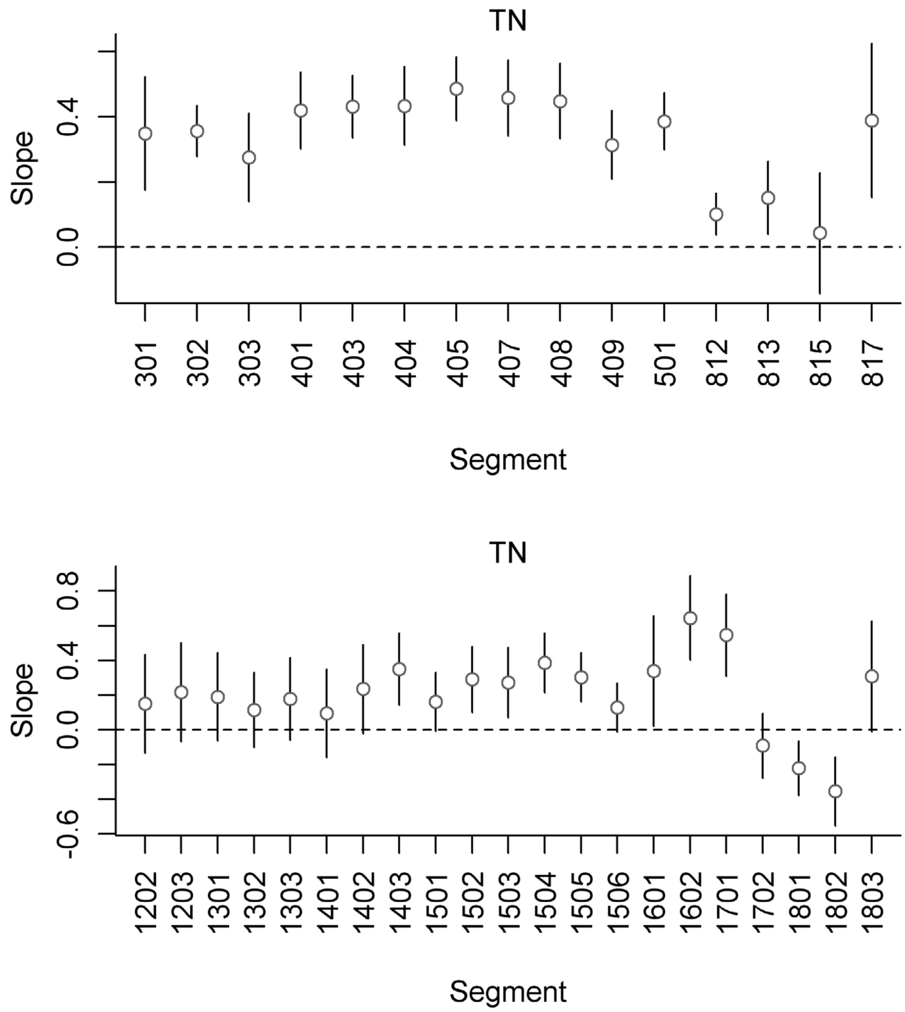


Figure 1-14. Estimates of the slope of the linear relationship between annual geometric mean TN and chl-a among different segments. Two panels from top to bottom show Northwest and Eastern estuaries. Open circles: mean estimate of slope; vertical lines: estimated 90% confidence intervals.

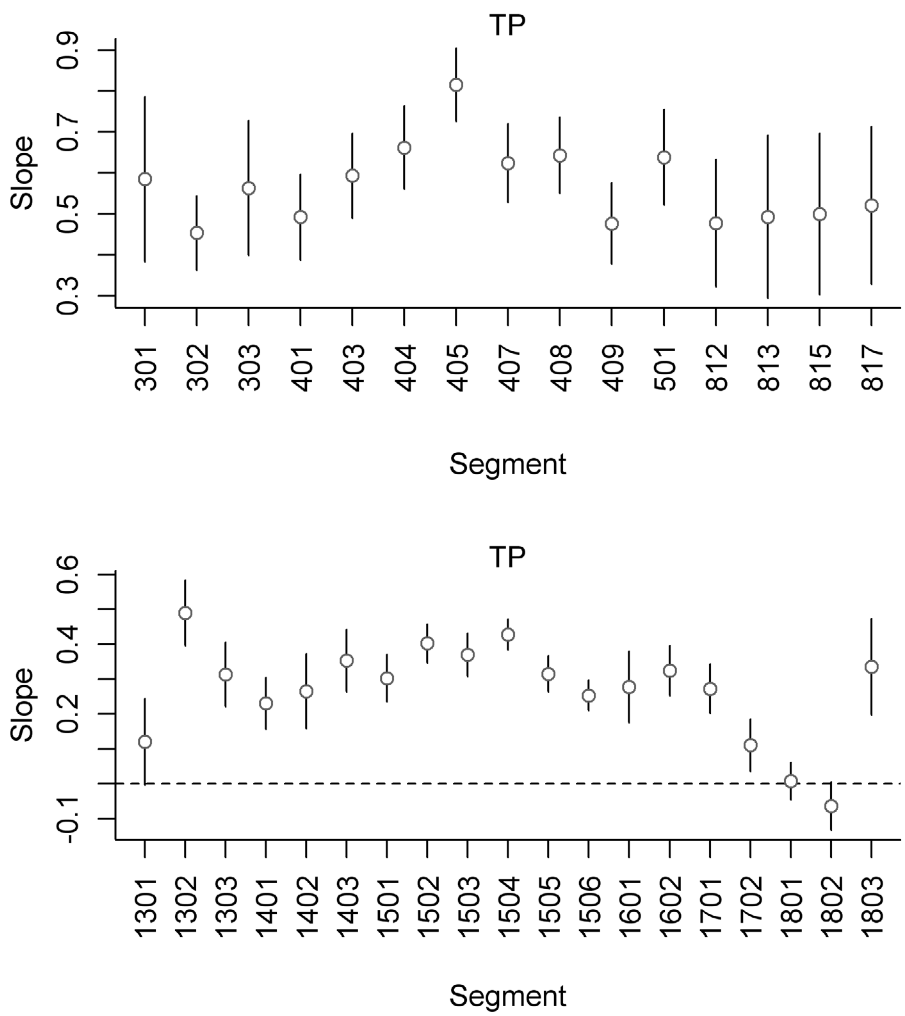


Figure 1-15. Estimates of the slope of the linear relationship between annual geometric mean TP and chl-a among different segments. Two panels from top to bottom show Northwest and Eastern estuaries. Open circles: mean estimate of slope; vertical lines: estimated 90% confidence intervals.

On balance, the observed relationships between nutrients and chl-a conform with expectations and provide a useful model for deriving numeric nutrient criteria.

1.4.3. Data Sources

Data sources for the empirical analysis and watershed and estuary modeling include monitoring data from numerous municipal, state, federal, and private sector sources, as summarized in Table 1-9.

Table 1-9. Data sources for empirical and water quality simulation modeling

Data	Source (Citation)
12-digit hydrologic unit (HUC12) watershed boundaries	U.S. Geological Survey (McFadden 1998; FDEP 2002; GSA No date; USGS No date a)
Climate data	Florida State Climatic Center (FSCC 2009)
Flow gaging stations	U.S. Geological Survey (USGS No date b)
Hydrologic group soils data	Natural Resource Conservation Service (NRCS No date)
IWR Run 40 water quality data	Florida Department of Environmental Protection (FDEP 2010)
Land use and impervious coverage	National Land Cover Dataset (Fry et al. 2011)
National Elevation Dataset (NED) digital elevation models	U.S. Geological Survey (USGS No date c)
National Hydrography Dataset (NHD) watershed boundaries and reaches	U.S. Geological Survey (USGS No date a)
Permit Compliance System municipal and industrial point sources	U.S. Environmental Protection Agency (USEPA No date)
Surface Airways Stations climate data	National Climatic Data Center (EarthInfo 2009)
Wastewater Facility Regulation municipal and industrial point sources	Florida Department of Environmental Protection (FDEP No date)
Atmospheric and wind data at Surface Airways Stations	National Climatic Data Center (EarthInfo 2009)
Bathymetric data	NOAA National Geophysical Data System (NOAA GEODAS No date)
Digitized shoreline data	NOAA (NOAA No date a)
IWR Run 40 salinity and temperature data	Florida Department of Environmental Protection (FDEP 2010)
Measured and Predicted Water Surface Elevation Data	NOAA (NOAA No date b)

1.5. Application of Analytical Approaches for Numeric Nutrient Criteria Derivation

EPA developed a systematic decision framework for derivation of numeric nutrient criteria. Using water quality simulation models and empirical/statistical model approaches as multiple lines of evidence, numeric criteria were derived by linking biological endpoints to concentrations of TN, TP, and chl-a in each estuary. In estuaries where sufficient monitoring data were available to statistically quantify relationships between TN, TP, chl-a, and biological endpoints, statistical models were used to derive the proposed numeric nutrient criteria. Where sufficient data were not available to apply statistical models (the stressor-response approach) in all segments in an estuary, EPA used mechanistic model simulation outputs to derive the criteria. In these instances, EPA analyzed the available stressor-response data as a second line of evidence in segments where the data were available.

The selection of the biological endpoints used to calculate criteria in each estuary was based on an evaluation of water quality simulation model sensitivity analyses of each endpoint, data availability, and analyses of monitoring data.

1.6. Analytical Approach Used to Derive Numeric Nutrient Criteria for the Protection of Downstream Estuaries

1.6.1. Introduction

Pursuant to 40 CFR part 131.10(b), water quality standards must ensure the attainment and maintenance of downstream water quality standards.[12] Thus, EPA is deriving numeric criteria for streams (i.e., flowing waters) in Florida in order to protect the estuarine water bodies that ultimately receive nitrogen and phosphorus pollution from the watershed. These criteria—which EPA refers to as DPVs—will apply in place of the stream's TN and TP criteria at the pour point if the applicable DPV is more stringent.

EPA is proposing a hierarchical procedure that includes four approaches for setting TN and TP DPVs. EPA's intention in proposing the four approaches is to provide a range of methods for the State to derive TN and TP DPVs that reflect the data and scientific information available. Water quality modeling is the most rigorous and most data-demanding method, and will generally result in the most refined DPVs. Water quality modeling is EPA's preferred method for establishing DPVs and is listed first in the hierarchy. It is followed by less rigorous methods that are also less data-demanding. Using a procedure from a lower tier of the hierarchy requires less data, but also generally results in more stringent DPVs to account for the uncertainties associated with these less refined procedures. The methods available to derive DPVs should be considered in the following order:

1. Water quality simulation models to derive TN and TP values,
2. Reference condition approach based on TN and TP concentrations at the stream pour point, coincident in time with the data record from which the downstream receiving estuary segment TN and TP criteria were developed using the same data quality screens and reference condition approach,
3. Dilution models based on the relationship between salinity and nutrient concentration in the receiving segment, and
4. The TN and TP criteria from the receiving estuary segment to which the freshwater stream discharges, in cases where data are too limited to apply the first three approaches.

EPA is proposing that DPVs for TN and TP be derived from estuary-specific application of water quality simulation models for estuaries where data and/or resources are available to configure and calibrate scientifically defensible estuary-specific water quality simulation models. EPA derived DPVs for 14 of the 23 estuarine systems in Florida according to this approach: Perdido Bay, Pensacola Bay, Choctawhatchee Bay, St. Joseph Bay, St. Andrews Bay, Apalachicola Bay, Alligator Harbor, Big Bend estuary systems, Lake Worth Lagoon, Loxahatchee River Estuary, St. Lucie River Estuary, St. Johns Estuary, Nassau River Estuary, and St. Marys River Estuary.

[12] 40 CFR part 131.10(b) reads "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."

In estuaries where data and/or resources are not available to configure and calibrate water quality models, EPA proposes the DPVs for TN and TP be derived from dilution models or the reference approach. EPA applied dilution models to develop DPVs for the Indian River Lagoon system, the Halifax River estuarine system, and the Guana, Tolomato, Matanzas, Pellicer (GTMP) estuarine system. Where a site specific reference condition approach is used to derive estuary criteria EPA proposes a reference approach for deriving DPVs. EPA applied this approach in South Florida where FDEP derived estuary and coastal criteria with a reference based approach (Subsection 62.302.532(1), F.A.C.).

In the absence of mechanistic models and sufficient empirical or reference data, the receiving estuary segment criteria would become the DPV. Florida derived numeric nutrient criteria for Clearwater Harbor/St. Joseph Sound, Tampa Bay, Sarasota Bay, and Charlotte Harbor/Estero Bay estuary systems. Absent analysis to calculate DPVs using mechanistic models, and sufficient empirical or reference data at the pour point, EPA is proposing Florida's in-estuary criteria as the applicable DPVs.

1.6.2. Analytical Approaches for DPV Derivation

The water quality modeling approach EPA is proposing for developing stream DPV criteria begins with estimates of limits on TN and TP loading rates needed to support the designated uses in estuaries (described in Section 1.4.1). Coupled watershed and hydrodynamic-water quality models were used to develop the protective TN and TP concentrations for estuary tributaries. EPA is proposing DPVs applicable within the terminal stream reaches or pour points to the estuary.

In Section 1.4.1, EPA described how watershed, hydrodynamic, and water quality models are used to simulate watershed and estuary processes to determine conditions that support estuary designated uses, and how these results are used to derive estuarine numeric nutrient criteria. For estuaries where water quality simulation models were developed, DPVs have been derived from the modeled stream loads entering the estuaries. DPVs were calculated at the stream pour point for the conditions that demonstrate attainment of the model endpoints. The model simulation that meets the endpoints represents water quality conditions that support designated uses for the estuary. The DPVs are either based on the calibrated model scenario or on nutrient reduction scenario, depending on the level of nutrients that result in meeting the endpoints.

In the reduction scenarios, watershed anthropogenic nutrient loads were reduced from calibrated levels until the estuary endpoints were met. Since most estuaries typically receive discharge from many tributary streams, a method of distributing the reductions among an estuary's tributaries had to be determined. EPA considered several methods as discussed in the 2010 Methods Document (USEPA 2010a) and is distributing the reductions in proportion to the estimate of natural and anthropogenic loading. For example, if the aggregate protective loading to the estuary is estimated to represent a 10 percent reduction in loading from anthropogenic sources, the loading distributed to each terminal reach would be based on 100 percent (no reduction) of the background loading from each tributary and a 10 percent reduction in the anthropogenic loading from each tributary. The background loading was estimated with the LSPC watershed model by simulating non-anthropogenic conditions in which all point sources of nutrients were set to zero, and nonpoint sources of nutrients were set to forest or wetland land use levels. Also, the non-anthropogenic proportion of load from a watershed changes daily, and this made it necessary to calculate the reduction for each day of the time series dataset. The resulting daily flows and concentrations were then entered into the EFDC and WASP estuary models, respectively. This method of reducing in proportion to the anthropogenic load contributed by each stream maintains the natural, background nutrient levels while reducing the controllable, anthropogenic part of the nutrient load. Finally, the DPVs were then calculated by computing annual geometric means for each of the modeled years 2002 through 2009 and calculating the 90^{th} percentile of these geometric means for each terminal tributary.

The dilution model approach was used to calculate DPVs where estuary criteria were derived using statistical models. In these areas, EPA developed dilution models based on the relationship between salinity and nutrient concentration recognizing that TN and TP follow conservative mixing principles, or dilution can be estimated from the estuarine salinity. As fresh water carrying nutrients pours into an estuary it mixes with the brackish estuarine water that has less nitrogen and phosphorus. As the water flows seaward it continues to mix and the nutrient concentration decreases and salinity increases. By plotting estuarine TN or TP criteria versus the average estuarine salinity and average sea TN or TP concentration versus average sea salinity and fitting a line to these two points the TN or TP at various levels of salinity can be determined. This analysis can then be used to determine the TN or TP concentration at the pour point that would provide for the attainment and maintenance of downstream estuarine waters. Predominantly fresh waters have been previously defined as surface waters in which the chloride concentration at the surface is less than 1,500 mg/L (salinity less than ~2.7 practical salinity unit [PSU]). Figure 1-16 below shows a schematic of the approach for developing the TN and TP relationship to salinity and then deriving the DPV. The DPV can be determined using the salinity associated with the pour point of 2.7 PSU and intersecting the line to find the DPV concentration associated with, and protective of, the estuarine waters where statistical models are applied. More detailed information can be found in Appendix B: Statistical (Stressor-Response) Analysis.

This approach results in a segment specific DPV that is applicable to all tributaries to that estuary segment. This approach cannot be used to allocate different nutrient concentrations between tributaries flowing into a single estuarine segment. This is a notable difference from the mechanistic modeling approach.

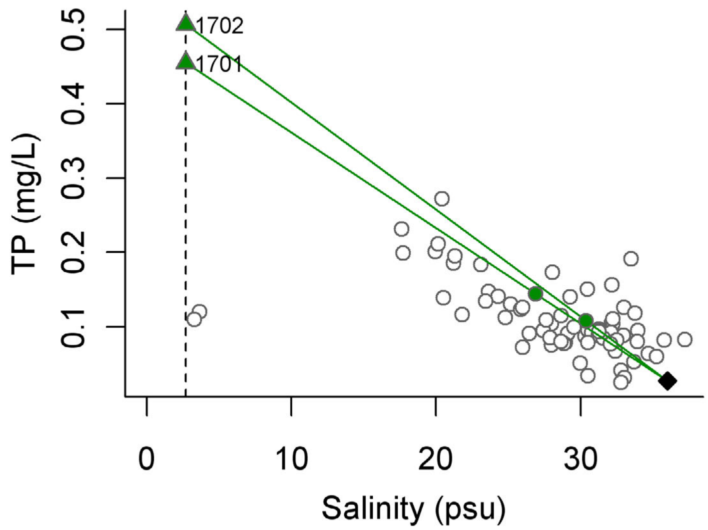


Figure 1-16. Dilution model approach schematic. Calculation of TP DPVs for GTMP. Black diamond shows seawater conditions, filled green circles show proposed TP criterion values for each segment, and green triangles show calculated DPVs. Open circles show observed long-term station average TP concentrations and salinities.

Another method applied only to south Florida marine waters (see Volume 3), the reference approach, was used to calculate DPVs where estuary criteria were derived using a reference condition based approach. In these areas, the 90th percentile annual geometric mean was calculated using observed water quality data near the pour point, which is similar to the way the downstream receiving marine segment's proposed TN and TP criteria were computed. Also, numeric DPVs computed in this approach reflect the same period of record over which the downstream receiving marine segment's TN and TP criteria were computed by FDEP. This way the DPV for the canal pour point is coincident with the estuary conditions used to derive the estuary criteria, and are protective of the estuary criteria. This approach results in a unique DPV for each tributary and reflects the current background and anthropogenic nutrient levels in the contributing watersheds.

EPA is proposing an additional approach for setting DPVs by applying the downstream receiving estuary segment criteria as the DPV. EPA applied this method to Clearwater Harbor/St. Joseph Sound, Tampa Bay, Sarasota Bay, and Charlotte Harbor/Estero Bay estuary systems. Such an approach may be appropriate where water quality simulation models and sufficient water quality data are not available to estimate the estuarine mixing and where estuarine systems are not of reference condition. This approach results in a segment specific DPV that is applicable to all tributaries to that estuary segment. This approach cannot be used to allocate different nutrient concentrations between tributaries flowing into a single estuarine segment. This is a notable difference from the mechanistic modeling approach.

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2. Estuary-Specific Numeric Nutrient Criteria and Downstream Protective Values

2.1. Perdido Bay

2.1.1. Proposed Numeric Nutrient Criteria

Proposed numeric nutrient criteria for TN, TP, and chl-a in Perdido Bay segments are summarized in Table 2-1.

Table 2-1. Proposed numeric nutrient criteria for Perdido Bay segments

Segment Name	Segment Number	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
Upper Perdido Bay	0101	0.59	0.042	5.2
Big Lagoon	0102	0.26	0.019	4.9
Central Perdido Bay	0103	0.47	0.031	5.8
Lower Perdido Bay	0104	0.34	0.023	5.8

2.1.2. General Characteristics

2.1.2.1. System Description

Perdido River and Perdido Bay (which is commonly divided into the upper, middle, and lower sections of the bay)[1] create the western border of Florida, with the state line bisecting the river and bay (FDEP 2010; Kirschenfeld et al. 2007). The bay itself covers about 50 mi^2 (130 km^2), but the entire basin is approximately 1,215 mi^2 (3,147 km^2), with 815 mi^2 (2,111 km^2) in Alabama and 400 mi^2 (1,036 km^2) in Florida (FDEP 2011; Macauley et al. 1995). Perdido River and its tributaries, Blackwater and Styx rivers, and Dyas and Brushy creeks, are the primary sources of freshwater to Perdido Bay. Elevenmile Creek (a small stream) and Bayou Marcus (a small bayou that drains urban runoff from a residential area) also feed into Perdido Bay (Grubbs and Pittman 1997; Livingston 2001).

On average, the Perdido area receives 63 in (160 cm) of rain per year. Late winter, spring, and summer are typically the wettest seasons, receiving precipitation from cold fronts in the winter and intense thunderstorms in the summer. Late spring and fall are typically the driest seasons (Grubbs and Pittman 1997). Between 1995 and 2009, 14 major tropical storms have made landfall near Perdido Bay (NOAA No date). Karst topography (typical of Florida) is not evident in the Perdido Basin. Soil erosion and stream sedimentation in the watershed occur because of highly erodible soils and hilly terrain in the area (FDEP 2008). Land use in the Perdido Bay

[1] The information presented in this system description was compiled to summarize local information pertaining to the proposed numeric nutrient criteria for this estuary. For more information on EPA's process of delineating, segmenting, and deriving numeric nutrient criteria for this estuary, please see Section 1.3.

watershed is approximately 50 percent forested or clear-cut, 18 percent agricultural, 15 percent forested wetlands, and 11 percent urban (Fry et al. 2011; NWFWMD 2009).[2]

The main bay changed from an oligohaline system to a saline system after an engineered pass to the Gulf of Mexico allowed more exchange between the bay and the gulf (FDEP 2010). Despite the low tidal range of 0.8 ft (0.2 m), tidal mixing plays an important role in the water quality of Perdido Bay (Grubbs and Pittman 1997; Xia et al. 2011). Upper Perdido Bay is influenced by freshwater inflow and tidal forcing, while Lower Perdido Bay is influenced mainly by tidal forcing (Xia et al. 2011).

2.1.2.2. Impaired Waters[3]

Four Class III marine water body identification numbers (WBIDs) in the Perdido Bay area have been listed for a nutrient-related parameter on Florida's CWA section 303(d) list approved by EPA. Of the four Class III marine WBIDs, two are impaired for DO (WBIDs 935 and 987), and two are impaired for nutrients (WBIDs 462A and 797). No Class II WBIDs with nutrient-related impairments are documented for this area.[4] No Class II or Class III marine nutrient-related total maximum daily loads (TMDLs) are documented for this region.[5]

2.1.2.3. Water Quality

All waters in the Perdido Bay Estuary are designated as Class III waters for recreation, propagation, and maintenance of a healthy, well-balanced population of fish and wildlife.[6] FDEP has designated several water bodies in the Perdido Bay system as Outstanding Florida Waters (OFWs).[7]

Factors that affect stratification and DO levels in Perdido Bay include tidal forcing and river discharge. Additionally, there are significant differences between winter and summer DO levels, especially in Upper Perdido Bay. In one study published in 2011, DO observations and modeling results showed that in Upper Perdido Bay bottom DO average concentrations were 1 mg/L in the summer, while surface waters differed by 8 mg/L (9 mg/L at the surface) in the summer. The difference between bottom and surface DO concentrations was approximately 1 mg/L in the winter. Lower Perdido Bay is influenced by tides, and in the same study, bottom DO

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[2] See Volume 1, Appendix C and its attachments for a summary of how land use data was combined and for a detailed breakdown of land uses.

[3] For more information about the data source, see Volume 1, Appendix A.

[4] The nutrient-related 303(d) list was a compilation of EPA-approved 303(d) listing information for nutrients, chl-a, and DO provided in three decision documents: September 2, 2009, Basin Groups 1, 2, and 5 EPA Decision Document (http://www.epa.gov/region4/water/tmdl/florida/documents/fl09303d_decisiondoc_090209.pdf);
May 13, 2010, Basin Group 3 EPA Decision Document
(http://www.epa.gov/region4/water/tmdl/florida/documents/fl303d_%20partialapproval_decision_docs051410.pdf);
and the December 21, 2010, Basin Group 4 EPA Decision Document
(http://www.epa.gov/region4/water/tmdl/florida/documents/group_4_final_dec_doc_and_partial_app_letter_12_21_10.pdf).

[5] TMDLs were identified in February 2011 by compiling nutrient-related draft/final TMDLs from the following three sources:
FDEP TMDL website (http://www.dep.state.fl.us/water/tmdl/index.htm)
EPA Region 4 website (http://www.epa.gov/region4/water/tmdl/florida/index.html)
EPA National WATERS expert query tool (http://www.epa.gov/waters/tmdl/expert_query.html)

[6] Section 62-302.400, F.A.C.

[7] Section 62-302.700, F.A.C.

concentrations were found to be more than 3 mg/L in the summer, with a small difference in bottom and surface water DO concentrations. In Lower Perdido Bay near Perdido Pass and the Gulf of Mexico, the waters are mixed more regularly by tidal forcing, and there is a lower vertical DO gradient, which is why hypoxic or anoxic conditions are rare (Xia et al. 2011).

Flemer and colleagues (1998) found that chl-a concentrations from May 1990 to April 1991 ranged from 0.5 to 10 µg/L in most samples collected in Upper Perdido Bay; from 3 to 12 µg/L in Middle Perdido Bay; and 2 to 10 µg/L in Lower Perdido Bay. FDEP reported that, for the entire Perdido Bay system, between 1995 and 2009 the geometric mean of chl-a concentrations for each year were between 1.1 and 6.0 µg/L (FDEP 2010). Water clarity was lower in Upper Perdido Bay (Secchi depth values of 0.7–4.3 ft [0.2–1.3 m]) than Lower Perdido Bay (Secchi depth values of 2.6–6.2 ft [0.8–1.9 m]). The authors also found that in Lower Perdido Bay salinity was generally higher than salinities measured in Upper Perdido Bay. Middle Perdido Bay salinities were generally intermediate of Upper and Lower Perdido Bay (Flemer et al. 1998). According to a different study conducted with water flow and water quality data collected between December 1994 and September 1995, solids concentrations measured at a cross section parallel to the U.S. Highway 98 Bridge in Perdido Bay were higher in September 1995 than in April 1995, whereas color was higher in April. Loads of total solids ranged from –12,300 to 14,800 lb/s (–5,600 to 6,700 kg/s) and suspended solids ranged from –90 to 185 lb/s (–41 to 84 kg/s) during 1995 sampling (Grubbs and Pittman 1997).[8]

2.1.2.4. Biological Characteristics

Florida Fish and Wildlife Conservation Commission (FFWCC) mapped vegetation and land cover for Florida using 2003 Landsat imagery (Stys et al. 2004). This mapping effort indicated that the majority of land cover vegetation within the Perdido Bay system was pineland; however, communities along the shoreline include coastal strand (contains vegetation such as grasses and shrubs), sand/beach (largely unvegetated), and salt marsh (typically made up of herbaceous and shrubby wetlands) (FDEP 2008; Gilbert and Stys 2004).

The three main species of seagrass found in Perdido Bay are widgeon grass (*Ruppia maritima*), shoal grass (*Halodule wrightii*), and turtle grass (*Thalassia testudinum*) (Kirschenfeld et al. 2007). The brackish and freshwater portions of the bay contain wild celery (*Vallisneria americana*), which tends to dominate Upper Perdido Bay (FDEP 2010). Lower Perdido Bay is dominated by shoal grass and turtle grass (Kirschenfeld et al. 2007). Between 1940 and 1992, 1.4 mi^2 (3.6 km^2 [877 ac]) (74%) of seagrasses were lost. Between 1992 and 2002, the rate of decline slowed but there was still a 0.01 mi^2 (0.03 km^2 [7 ac]) (2.6%) loss (Kirschenfeld et al. 2007). The majority of the seagrass loss was in Lower Perdido Bay where there was a decrease from 1.7 mi^2 (4.3 km^2 [1,072 ac]) in 1940 to 0.47 mi^2 (1.2 km^2 [299 ac]) in 2002. The loss in Lower Perdido Bay is associated with accelerated development of residential, resort, and marine areas. In Upper and Middle Perdido Bay, seagrass decline can be attributed to high nutrient inputs from wastewater and paper mill point sources, as well as nonpoint source runoff from agricultural, silvicultural, and residential lands (Kirschenfeld et al. 2007).

[8] Positive load values indicate a net-seaward load, while negative load values indicate a net-landward load.

Algal blooms in Perdido Bay are dominated by three species: *Cyclotella choctawhatcheeana* (a diatom), *Prorocentrum cordatum* (a dinoflagellate), and *Heterosigma akashiwo* (a raphidophyte) (Livingston 2007).

No waters in the Perdido Bay basin are designated for shellfish propagation, and there are no open, active shellfish-harvesting areas (FDEP 2008). After phytoplankton blooms in 1993–1994, the macroinvertebrate population declined (Livingston 2001). The Perdido Bay system provides important habitat for many species, including recreationally and commercially important fish species. Salt marsh topminnow and Atlantic sturgeon also use the bay and its tributaries for spawning, as a nursery, and as adult habitat (FFWCC 2012).

For a more detailed summary of this water body, see Volume 1, Appendix A.

2.1.3. Data Used

Several data sources specific to Perdido Bay were used in addition to those sources described in Section 1.4.3, as summarized in Table 2-2.

Table 2-2. Data sources specific to Perdido Bay models

Data	Source	Location Used
Municipal and industrial point sources	Alabama Department of Environmental Management (ADEM No date a, No date b)	Perdido watershed model
Water quality data	Alabama Department of Environmental Management (ADEM 2006)	Perdido watershed model
FDEP Level III Florida Land Use	Northwest Florida Water Management District (NWFWMD 2009)	Perdido watershed model

2.1.4. Segmentation

Perdido Bay was segmented into Upper, Central, and Lower Perdido Bay in a seagrass assessment study conducted by Kirschenfeld et al. (2007). The Geographic Information System (GIS) isohaline analysis yielded a similar segmentation scheme. The lower portion of the bay was further subdivided into two parts in accordance with the salinity gradient. Further, the portion of the bay in Alabama was removed by truncating the segments at the Alabama-Florida state line. Figure 2-1 shows the resulting four segments for Perdido Bay.

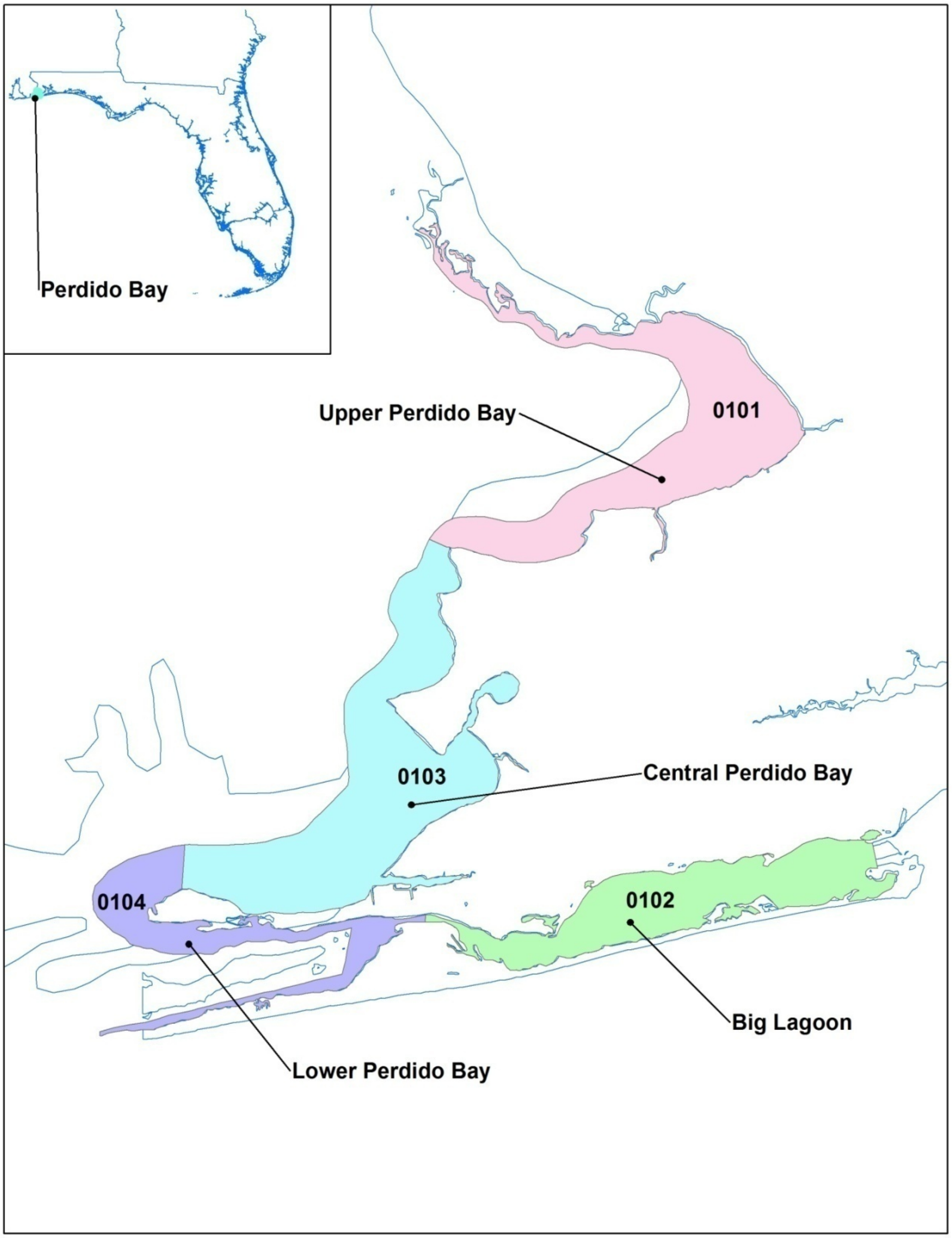

Figure 2-1. Results of Perdido Bay segmentation

2.1.5. Water Quality Targets

2.1.5.1. Seagrass Depth and Water Clarity Targets

Seagrass depth of colonization (measured as Z_c) and water clarity (measured as K_d) targets were established for two of the four Perdido Bay estuary segments after evaluating seagrass coverage from 1940, 1992, and 2003 (Table 2-3) (USGS 2012). The deepest depth of colonization was for the 1940 coverage, which was used to compute a target for segment 0104. The 1960 Pensacola Bay seagrass coverage, which included coverage for Big Lagoon, was evaluated to determine a depth of colonization target for segment 0102. Both targets were consistent with average carbonaceous dissolved organic matter (CDOM) in the segment. Figure 2-2 shows seagrass distribution in Perdido Bay in 1940.

Table 2-3. Perdido Bay seagrass depth and water clarity targets by segment

Estuary Segment	Depth of Colonization (Z_c) Target (m)	Light Attenuation Coefficient (K_d) Target (1/m)
0101	No target	-
0102	1.7	0.9
0103	No target	-
0104	1.9	0.8

Figure 2-2. Seagrass distribution in Perdido Bay in 1940

2.1.5.2. Chlorophyll a Target

To prevent nuisance algal blooms and protect the estuary's designated uses, chl-a levels must not exceed 20 µg/L more than 10 percent of the time. The rationale for that target is provided in Section 1.2.2.

2.1.5.3. Dissolved Oxygen Targets

On the basis of the rationale that sufficient DO is necessary to protect aquatic life, as described in Section 1.2.3, the following DO targets were established:

- Minimum allowable DO of 4.0 mg/L as a water column average for each estuary segment 90 percent of the time over the simulation's time span
- Daily average DO of 5.0 mg/L as a water column average for each estuary segment 90 percent of the time over the simulation's time span
- Minimum 3-hour average DO of 1.5 mg/L in the bottom two layers for each estuary segment over the simulation's time span

2.1.6. Results of Analyses

2.1.6.1. Mechanistic Model Analysis

Average load contributions from the Perdido watershed are shown in Table 2-4.

Table 2-4. Average load contributions from the Perdido watershed (2002–2009)

	TN Load[1] (kg/d; Mean ± se)			TP Load[1] (kg/d; Mean ± se)		
Year	Existing[2]	Background[3]	Anthropogenic[4]	Existing[2]	Background[3]	Anthropogenic[4]
2002	3,347 ± 174	1,116 ± 90	2,231 ± 86	209 ± 7	20 ± 4	189 ± 3
2003	5,540 ± 257	2,345 ± 152	3,195 ± 111	270 ± 10	45 ± 5	225 ± 5
2004	4,340 ± 153	1,611 ± 80	2,729 ± 78	238 ± 7	26 ± 3	212 ± 4
2005	5,838 ± 344	2,322 ± 161	3,516 ± 193	263 ± 12	39 ± 5	224 ± 8
2006	2,338 ± 64	696 ± 28	1,642 ± 39	173 ± 4	11 ± 1	161 ± 3
2007	2,689 ± 115	783 ± 47	1,905 ± 72	228 ± 6	12 ± 2	216 ± 4
2008	3,687 ± 145	1,270 ± 67	2,417 ± 82	218 ± 6	19 ± 2	199 ± 5
2009	5,312 ± 225	1,955 ± 100	3,357 ± 127	254 ± 8	32 ± 3	222 ± 5

[1] Annual average daily load from all terminal reaches
[2] Annual average daily load from all terminal reaches computed from existing model scenario
[3] Annual average daily load from all terminal reaches computed from background model scenario (remove anthropogenic sources/land use; use existing hydrology)
[4] Anthropogenic load computed as difference between existing and background load

For Perdido Bay, DO and chl-a targets were met for each segment on the basis of 2002–2009 nutrient loads. The light attenuation coefficient target was not met for segment 0104 under either the 2002–2009 nutrient loads or non-anthropogenic nutrient scenario. Table 2-5 identifies which targets were met under 2002–2009 nutrient loads and which targets were not met.

An evaluation of model sensitivity to the water quality targets applied revealed that all targets were sensitive to nutrients changes in Perdido Bay.

Table 2-5. Water quality targets met for Perdido Bay based on mechanistic modeling

Segment	DO	Chl-a	K_d
0101	Yes	Yes	No target
0102	Yes	Yes	Yes
0103	Yes	Yes	No target
0104	Yes	Yes	No

A summary of candidate criteria based on the 2002–2009 nutrient loads for Perdido Bay Estuary segments is given in Table 2-6.

Table 2-6. Summary of candidate criteria for Perdido Bay derived from mechanistic modeling

Segment	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
0101	0.59	0.042	5.2
0102	0.26	0.019	4.9
0103	0.47	0.031	5.8
0104	0.34	0.023	5.8

2.1.6.2. Statistical Model Analysis

Data were not sufficient within each segment to conduct statistical analyses in Perdido Bay.

2.1.7. Application of Analyses for Proposed Numeric Nutrient Criteria

Although data necessary to conduct statistical analyses were not available for all segments of Perdido Bay, the mechanistic model provided values for every segment in the estuary. As a result, EPA derived the proposed numeric nutrient criteria for Perdido Bay based on the mechanistic modeling results.

EPA evaluated three endpoints in the mechanistic modeling approach for Perdido Bay: (1) water clarity protective of historic depth of seagrasses, (2) chl-a concentrations associated with balanced phytoplankton biomass, and (3) DO concentrations sufficient to maintain aquatic life. EPA found that all three targets were achieved under the calibrated 2002–2009 nutrient loads. EPA also found that all three endpoints were sensitive to changes in nutrients. The proposed criteria were derived to be protective of water clarity, DO, and chl-a concentrations. The values under mechanistic modeling represent the 90th percentile annual geometric mean nutrient concentrations from the 2002–2009 modeled nutrient loads.

Proposed numeric nutrient criteria for TN, TP, and chl-a in Perdido Bay segments are summarized in Table 2-7.

Table 2-7. Proposed and candidate numeric nutrient criteria for Perdido Bay segments

		Proposed Criteria			Mechanistic Modeling		
SEGMENT	SEGMENT ID	TN (mg/L)	TP (mg/L)	Chl-a (μg/L)	TN (mg/L)	TP (mg/L)	Chl-a (μg/L)
Upper Perdido Bay	0101	0.59	0.042	5.2	0.59	0.042	5.2
Big Lagoon	0102	0.26	0.019	4.9	0.26	0.019	4.9
Central Perdido Bay	0103	0.47	0.031	5.8	0.47	0.031	5.8
Lower Perdido Bay	0104	0.34	0.023	5.8	0.34	0.023	5.8

2.1.8. Downstream Protective Values

In Perdido Bay mechanistic models were applied to derive the proposed DPVs for TN and TP shown in Table 2-8.

Table 2-8. Proposed DPVs for Perdido Bay

Tributary[a]	LSPC Model Watershed ID	USGS Reach Code	Estuary Segment	TN (mg/L)	TP (mg/L)
Bridge Creek	10006	03140107000130	0101	0.72	0.005
Bayou Marcus	10011	03140107002683	0101	1.13	0.041
	10012	03140107002610	0104	0.85	0.005
	10013	03140107000174	0103	0.58	0.006
	10015	03140105002337	0103	0.73	0.006
	10050	03140107002231	0102	1.11	0.006
	10053	03140107002452	0102	0.83	0.005
Perdido River	10060	03140106000002	0101	0.55	0.770
	10062	03140107000741	0101	0.04	0.005
Elevenmile Creek	10066	03140107002622	0101	2.18	0.216

[a] Tributary names left blank are unnamed

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2.2. Pensacola Bay

2.2.1. Proposed Numeric Nutrient Criteria

Proposed numeric nutrient criteria for TN, TP, and chl-a in Pensacola Bay segments are summarized in Table 2-9.

Table 2-9. Proposed numeric nutrient criteria for Pensacola Bay segments

Segment Name	Segment Number	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
Blackwater Bay	0201	0.53	0.022	3.9
Upper Escambia Bay	0202	0.43	0.025	3.7
East Bay	0203	0.50	0.021	4.2
Eastern Santa Rosa Sound	0204	0.34	0.018	4.1
Lower Escambia Bay	0205	0.44	0.023	4.0
Upper Pensacola Bay	0206	0.40	0.021	3.9
Lower Pensacola Bay	0207	0.34	0.020	3.6
Western Santa Rosa Sound	0208	0.33	0.020	3.9
Central Santa Rosa Sound	0209	0.36	0.020	4.9

2.2.2. General Characteristics

2.2.2.1. System Description

The Pensacola Bay system is along the western Florida Panhandle. Freshwater from the Escambia, Blackwater, and Yellow rivers feeds the system, with the northwestern portion receiving three times more freshwater than the northeastern portion (USEPA 2005). The watershed covers roughly 7,000 mi^2 (18,130 km^2), with about 550 mi (885 km) of coastline and 144 mi^2 (373 km^2) of surface waters (FDEP 2010; Schwenning et al. 2007; Thorpe et al. 1997). The majority of the watershed is in Alabama (about 65%); the rest is in Florida (Schwenning et al. 2007). The entire system drains into the Gulf of Mexico, primarily through a half-mile-wide (1 km) channel at the southwest corner of the bay (FDEP 2010; Thorpe et al. 1997). Pensacola Bay has five distinct sections: Escambia Bay, East Bay, Santa Rosa Sound, Pensacola Bay, and Big Lagoon (Schwenning et al. 2007).[9]

The Pensacola Bay system experiences a subtropical climate. According to the 30-year average, the wettest months are generally June to September with an average precipitation of 5.8–8.0 in (14.7–20.3 cm) per month of rain, and the driest are October to December with an average precipitation of 4.0–4.5 in (10.2–11.4 cm) per month (NWFWMD 2011). The average depth of the system is 7.6 ft (2.3 m) in the northern areas and 19.7 ft (6.0 m) in the southern areas. The deepest parts of the bay, about 33–66 ft (10–20 m), were dredged to create a channel for navigational purposes (USEPA 2005). Minimal tidal exchange occurs with the Gulf of Mexico through the pass at the southwestern part of the bay system. Average tidal range is about 1.6 ft (0.5 m). It takes about 21 to 34 days to flush the system (USEPA 2005).

Major land use types in the watershed are clear-cut or forested (over 50%), forested wetlands (11%), and developed land (9%) (Fry et al. 2011; NWFWMD 2009).[10]

Much of the developed land in the Pensacola Bay watershed is concentrated in the immediate vicinity of Pensacola Bay. The most extensive urban, industrial, commercial and residential development is along the western shores of the bay in Escambia County. Lower density, mainly residential, development surrounds much of the rest of the bay, particularly on the Gulf Breeze Peninsula and on Santa Rosa Island. Farther from the bay, land use in Escambia County quickly changes to forestry with some row crop agriculture. In the eastern portion of the watershed, Eglin Air Force Base occupies more than 734 mi^2 (1,902 km^2 [470,000 ac]). Blackwater River State Forest and Wildlife Management Area constrains urban development and is managed for conservation and recreation (Thorpe et al. 1997).

2.2.2.2. Impaired Waters[11]

Eight Class II and Class III marine WBIDs in the Pensacola Bay system are listed for a nutrient-related parameter on Florida's CWA section 303(d) list approved by EPA. Of the eight WBIDs, two are Class II WBIDs, and six are Class III marine WBIDs. The two Class II WBIDs are impaired for DO (WBIDs 548C and 701A). Of the six Class III marine WBIDs, two are impaired for DO (WBIDs 10F and 740), one is impaired for nutrients and DO (WBID 493), one is

[9] The information presented in this system description was compiled to summarize local information pertaining to the proposed numeric nutrient criteria for this estuary. For more information on EPA's process of delineating, segmenting, and deriving numeric nutrient criteria for this estuary, please see Section 1.3.

[10] See Volume 1, Appendix C and its attachments for a summary of how land use data was combined and for a detailed breakdown of land uses.

[11] For more information about the data source, see Volume 1, Appendix A.

impaired for nutrients and chl-a (WBID 846), and two are impaired for nutrients, chl-a, and DO (WBIDs 548AA and 846C).[12]

One draft nutrient-related TMDL for Class II or Class III marine WBIDs exists in the Pensacola Bay watershed, the draft *Escambia Bay Nutrient TMDL,* covering WBIDs 548AA, 548AB, and 548AC.[13]

2.2.2.3. Water Quality

The extent of hypoxia (DO concentrations ≤ 2 mg/L) in the bay increases during the summer. Approximately 24 percent of the bay system has DO levels less than 2.0 mg/L in summer (USEPA 2005). Hypoxia was particularly severe during summer 2004, with many DO measurements below 1 mg/L. Average summer bottom DO levels from 2000 to 2004 reached a minimum of just below 2.0 mg/L (Hagy et al. 2006). Despite physical characteristics not typically associated with hypoxia (i.e., being broad and shallow), the extent and frequency of hypoxia suggests that portions of the bay are prone to developing hypoxia. Strong density stratification, low rates of vertical mixing, and weak estuarine circulation in summer months contribute to hypoxia (Hagy and Murrell 2007). Livingston (2006) reported that bottom DO in upper and mid-Escambia Bay and the Blackwater–East Bay system was controlled by salinity stratification, but that a relationship was not seen in Pensacola Bay.

According to EPA's *Ecological Condition Report of the Pensacola Bay System, Northwest Florida* (2005), nearly half of all chl-a measurements were below 5 µg/L, whereas the remaining were between 5 and 20 µg/L (USEPA 2005). Chl-a concentrations generally tracked phytoplankton productivity, with the highest concentrations during summer and the lowest during winter (Murrell et al. 2007). EPA's *Ecological Condition Report of the Pensacola Bay System, Northwest Florida* reported water clarity to be poor when less than 10 percent of the ambient light is observed at a depth of 3.3 ft (1 m). With that guideline, less than 10 percent of the bay system was considered to have poor water clarity, with the greatest extent of poor conditions occurring during the summer. In addition to suspended solids, CDOM also affects light penetration and is elevated in upper areas of Pensacola Bay, such as Blackwater Bay (USEPA 2005).

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[12] The nutrient-related 303(d) list was a compilation of EPA-approved 303(d) listing information for nutrients, chl-a, and DO provided in three decision documents: September 2, 2009, Basin Groups 1, 2, and 5 EPA Decision Document (http://www.epa.gov/region4/water/tmdl/florida/documents/fl09303d_decisiondoc_090209.pdf);
May 13, 2010, Basin Group 3 EPA Decision Document
(http://www.epa.gov/region4/water/tmdl/florida/documents/fl303d_%20partialapproval_decision_docs051410.pdf);
and the December 21, 2010, Basin Group 4 EPA Decision Document
(http://www.epa.gov/region4/water/tmdl/florida/documents/group_4_final_dec_doc_and_partial_app_letter_12_21_10.pdf).

[13] TMDLs were identified in February 2011 by compiling nutrient-related draft/final TMDLs from the following three sources:
FDEP TMDL website (http://www.dep.state.fl.us/water/tmdl/index.htm)
EPA Region 4 website (http://www.epa.gov/region4/water/tmdl/florida/index.html)
EPA National WATERS expert query tool (http://www.epa.gov/waters/tmdl/expert_query.html)

2.2.2.4. Biological Characteristics

Marsh habitat is found in the tidal creeks and lower reaches of river floodplains in the bay system. In areas with higher salinity, black needlerush (*Juncus roemerianus*) and salt marsh cordgrass (*Spartina alterniflora*) are common species of marsh vegetation, and sawgrass (*Cladium jamaicense*), pickerelweed (*Pontederia lanceolata*), bulrush (*Scirpus validus*), and cattail (*Typha* spp.) dominate areas with lower salinity (FDEP 2010). The dominant seagrass species in the Pensacola Bay system are shoal grass (*Halodule wrightii*) and turtle grass (*Thalassia testudinum*) (Schwenning et al. 2007; USGS 2001). Other species include manatee grass (*Syringodium filiforme*), star grass (*Halophila engelmannii*), and widgeon grass (*Ruppia maritime*) (FDEP 2010; Schwenning et al. 2007).

During the 1960s, approximately 15 mi^2 (38 km^2 [9,500 ac]) of seagrass were observed in the entire Pensacola Bay system. By 1992 that had decreased to 7.0 mi^2 (18 km^2 [4,500 ac]). USGS reported that the major causes for seagrass decline in Pensacola Bay were sewage and industrial waste discharges, dredge and fill activities, and beachfront alterations (USGS 2001). By 2003 seagrasses in Pensacola Bay, East Bay, and Escambia Bay covered 0.80 mi^2 (2.07 km^2 [511 ac]), a 43 percent decline from 1992. That loss was likely caused by increased salinity in the upper reaches of Escambia Bay and East Bay, where the habitat loss occurred. In Santa Rosa Sound, seagrasses covered 4.7 mi^2 (12 km^2 [3,032 ac]) in 2003, which was a 9.9 percent increase from 1992. The new seagrasses identified in 2003 in Santa Rosa Sound are primarily turtle grass and some shoal grass (FFWCC 2011).

Murrell et al. (2007) found in their 1999–2001 study that higher phytoplankton production rates were associated with freshwater flow and flushing times in Escambia Bay. Phytoplankton community composition in the bay changes seasonally, likely in association with water temperature. For example, in a study conducted from 1999 to 2001, cyanobacteria abundance was strongly correlated to water temperature (Murrell and Lores 2004).

Common invertebrate species in the bay system include blue crab, American oyster, and Penaeid shrimp (FDEP 2010). Portions of the Pensacola Bay system are used for shellfish harvesting (FDACS 2004a, 2004b). Pensacola Bay was once known for its thriving oyster industry, but by 1971 over 90 percent of Escambia Bay's commercially harvestable oysters were found dead from the fungus *Perkinsus marina* (formerly called *Labyrinthomyxa marina*). Because of the lack of suitable substrate and disease, the oysters have been slow to recover (USEPA 2005). The highest concentrations of polychaete worms, brown shrimp, and blue crabs were found in the upper Escambia Bay and Blackwater Bay between May 1997 and October 1998. During the same period, relatively low biomass of macroinvertebrates and infauna were found in eastern sections of upper Escambia Bay and in lower Escambia Bay. Indicator species for pollution were also found. Compared to similar alluvial, river-dominated estuaries (e.g., Perdido), the Pensacola Bay system has relatively low overall secondary productivity (Livingston 2006).

More than 200 species of fish and shellfish have been identified in estuarine waters of the Pensacola Bay system. Common species include spot, bay anchovy, Atlantic croaker, spotted seatrout, gulf menhaden, and striped mullet. Largemouth bass and redear sunfish are freshwater species that can tolerate low salinities. Gulf sturgeon, Alabama shad, skipjack herring, and Gulf Coast striped bass are anadromous species that use the bay system and its tributaries (FDEP 2010). From 1993 to 1998 a study collected about 585,000 fish from three urban Pensacola Bay bayous. These fish were classified as belonging to 80 species, 66 genera, and 37 families. During the study period the community structure remained seasonably stable, even during two hurricanes. Many of the collected species have shown resilience to fluctuating conditions (i.e., DO levels of 1.2–14.2 mg/L) (Lewis et al. 2011).

For a more detailed summary of this water body, see Volume 1, Appendix A.

2.2.3. Data Used

Several data sources specific to Pensacola Bay were used in addition to those sources described in Section 1.4.3, as summarized in Table 2-10.

Table 2-10. Data sources specific to Pensacola Bay models

Data	Source (Citation)	Location Used
Municipal and industrial point sources	Alabama Department of Environmental Management (ADEM No date a, No date b)	Pensacola watershed model
Water quality data	Alabama Department of Environmental Management (ADEM 2006)	Pensacola watershed model
FDEP Level III Florida Land Use	Northwest Florida Water Management District (NWFWMD 2009)	Pensacola watershed model
Continuous Record of Salinity and Water Temperature in Pensacola Bay	U.S. Environmental Protection Agency Gulf Ecology Division (Hagy and Murrell 2007)	Pensacola estuary model
Monthly Water Quality Data in Pensacola Bay 1998-2004	U.S. Environmental Protection Agency Gulf Ecology Division (Hagy and Murrell 2007)	Pensacola estuary model

2.2.4. Segmentation

The results of GIS isohaline analysis yielded nine segments for Pensacola Bay. The resulting segmentation was similar to the one proposed by Hagy and Murrell (2007). The Santa Rosa Sound segment was further subdivided into three distinct segments to account for the variation in seagrass depth distribution in Santa Rosa Sound. Figure 2-3 shows the resulting nine segments for Pensacola Bay.

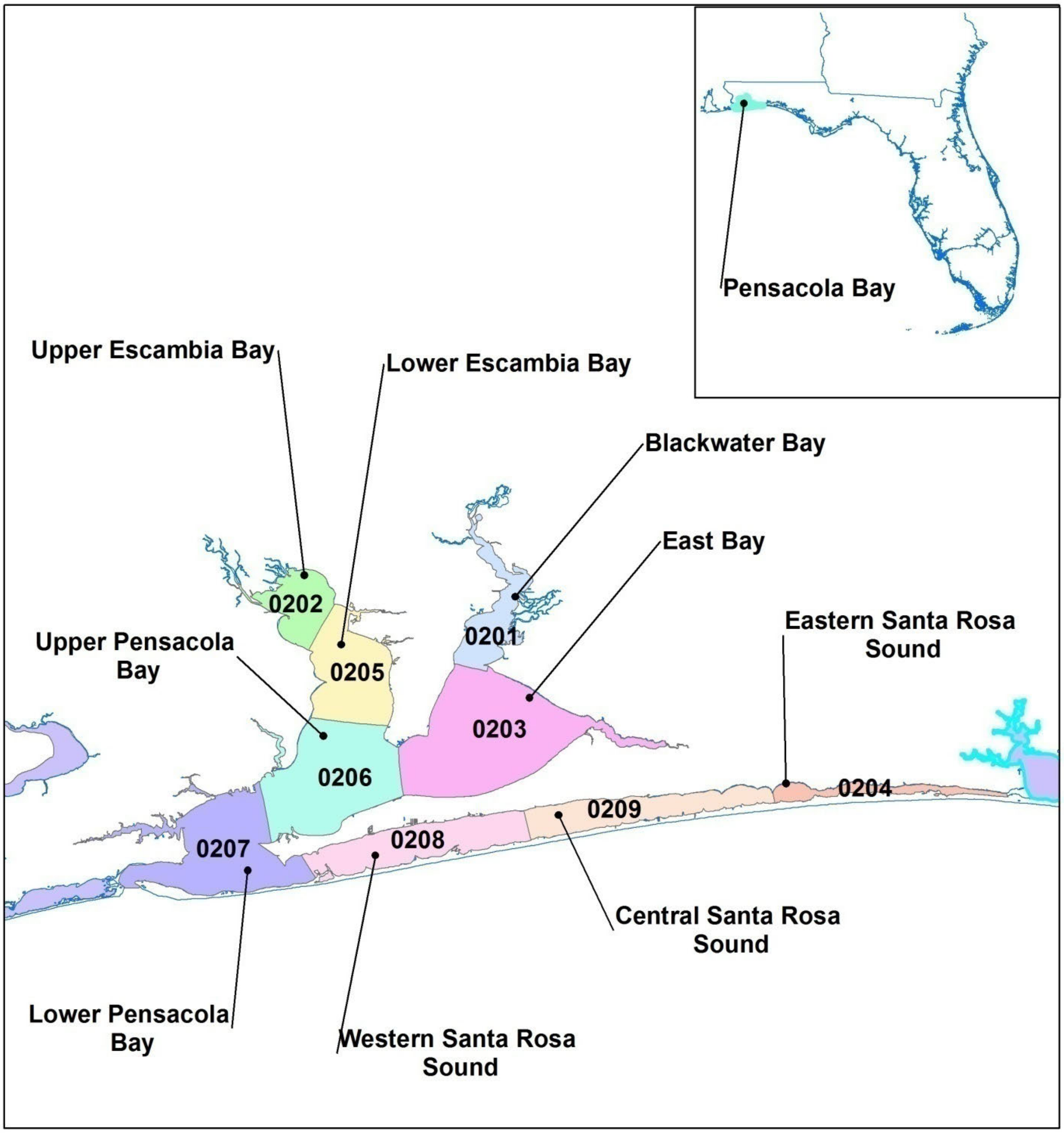

Figure 2-3. Results of Pensacola Bay segmentation

2.2.5. Water Quality Targets

2.2.5.1. Seagrass Depth and Water Clarity Targets

Seagrass depth of colonization (measured as Z_c) and water clarity (measured as K_d) targets were established for the Pensacola Bay estuary after evaluating the 1960, 1992, 2003, and 2010 seagrass coverages (Table 2-11) (USGS 2012). The maximum depth of colonization in each segment was observed from the 1960 seagrass coverage (Figure 2-4), which was used as the target. Targets were found to be consistent with average CDOM.

Table 2-11. Pensacola Bay seagrass depth and water clarity targets by segment

Estuary Segment	Depth of Colonization (Z_c) Target (m)	Light Attenuation Coefficient (K_d) Target (1/m)
0201	0.70	2.3
0202	0.60	2.7
0203	1.10	1.5
0204	1.70	0.9
0205	0.70	2.3
0206	1.40	1.1
0207	2.40	0.7
0208	2.70	0.6
0209	1.80	0.9

Figure 2-4. Seagrass coverage for Pensacola Bay in 1960

2.2.5.2. Chlorophyll a *Target*

To prevent nuisance algal blooms and protect the estuary's designated uses, chl-a levels must not exceed 20 µg/L more than 10 percent of the time. The rationale for that target is provided in Section 1.2.2.

2.2.5.3. Dissolved Oxygen Targets

On the basis of the rationale that sufficient DO is necessary to protect aquatic life, as described in Section 1.2.3, the following DO targets were established:

- Minimum allowable DO of 4.0 mg/L as a water column average for each estuary segment 90 percent of the time over the simulation's time span
- Daily average DO of 5.0 mg/L as a water column average for each estuary segment 90 percent of the time over the simulation's time span
- Minimum 3-hour average DO of 1.5 mg/L in the bottom two layers for each estuary segment over the simulation's time span

2.2.6. Results of Analyses

2.2.6.1. Mechanistic Model Analysis

Average load contributions from the Pensacola watershed are shown in Table 2-12.

Table 2-12. Average load contributions from the Pensacola watershed (2002–2009)

	TN Load[1] (kg/d; Mean ± se)			TP Load[1] (kg/d; Mean ± se)		
Year	Existing[2]	Background[3]	Anthropogenic[4]	Existing[2]	Background[3]	Anthropogenic[4]
2002	13,199 ± 605	6,559 ± 337	6,640 ± 272	728 ± 37	288 ± 24	440 ± 16
2003	25,297 ± 928	13,398 ± 563	11,900 ± 383	1,211 ± 51	516 ± 37	694 ± 18
2004	24,777 ± 1,098	13,392 ± 718	11,385 ± 397	1,434 ± 105	746 ± 88	688 ± 22
2005	33,107 ± 2,043	18,129 ± 1,199	14,978 ± 866	1,668 ± 132	888 ± 104	780 ± 35
2006	8,263 ± 323	3,973 ± 169	4,290 ± 157	481 ± 21	168 ± 12	313 ± 10
2007	11,808 ± 507	4,913 ± 261	6,895 ± 250	631 ± 29	208 ± 17	423 ± 13
2008	21,087 ± 1,060	10,045 ± 634	11,043 ± 443	1,097 ± 80	441 ± 61	656 ± 24
2009	29,231 ± 1,293	14,365 ± 713	14,866 ± 587	1,452 ± 73	569 ± 47	883 ± 30

[1] Annual average daily load from all terminal reaches
[2] Annual average daily load from all terminal reaches computed from existing model scenario
[3] Annual average daily load from all terminal reaches computed from background model scenario (remove anthropogenic sources/land use; use existing hydrology)
[4] Anthropogenic load computed as difference between existing and background load

For Pensacola Bay, the chl-a target was met for each segment on the basis of 2002–2009 nutrient loads. The DO targets were met on the basis of 2002–2009 nutrient loads, except for the daily water column average target of 5 mg/L, which could not be met for segments 0206 and 0208. Reduction runs were required to meet the DO targets. The light attenuation coefficient target was not met for segments 0208 and 0209 under either the 2002–2009 nutrient loads or non-anthropogenic nutrient scenario. Table 2-13 identifies which targets were met under 2002–2009 nutrient loads and which targets were not met.

An evaluation of model sensitivity to the water quality targets applied revealed that the water clarity target was insensitive to nutrients changes in Pensacola Bay.

Table 2-13. Water quality targets met for Pensacola Bay based on mechanistic modeling

Segment	DO	Chl-a	K_d
0201	Yes	Yes	Yes
0202	Yes	Yes	Yes
0203	Yes	Yes	Yes
0204	Yes	Yes	Yes
0205	Yes	Yes	Yes
0206	No	Yes	Yes
0207	Yes	Yes	Yes
0208	No	Yes	No
0209	Yes	Yes	No

A summary of candidate criteria for Pensacola Bay segments, based on a reduction of the 2002–2009 nutrient loads to meet the DO target is given in Table 2-14.

Table 2-14. Summary of candidate criteria for Pensacola Bay derived from mechanistic modeling

Segment	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
0201	0.53	0.022	3.9
0202	0.43	0.025	3.7
0203	0.50	0.021	4.2
0204	0.34	0.018	4.1
0205	0.44	0.023	4.0
0206	0.40	0.021	3.9
0207	0.34	0.020	3.6
0208	0.33	0.020	3.9
0209	0.36	0.020	4.9

2.2.6.2. Statistical Model Analysis

Data were not sufficient within each segment to conduct statistical analyses in Pensacola Bay.

2.2.7. Application of Analyses for Proposed Numeric Nutrient Criteria

Data necessary to conduct statistical analyses were not available for all segments of Pensacola Bay. However, the mechanistic model provided values for every segment in the estuary. As a result, EPA derived the proposed numeric nutrient criteria for Pensacola Bay based on the mechanistic modeling results.

EPA evaluated three endpoints in the mechanistic modeling approach for Pensacola Bay: (1) water clarity protective of historic depth of seagrasses, (2) chl-a concentrations associated with balanced phytoplankton biomass, and (3) DO concentrations sufficient to maintain aquatic life. EPA found that the water clarity target was not met under 2002–2009 loads and was insensitive to changes in nutrients. Therefore, the water clarity endpoint was not used in Pensacola Bay. The DO target was also not achieved with the 2002–2009 nutrient loads, but was shown to be sensitive to changes in nutrients. The chl-a target was met and was also shown to be sensitive to changes in nutrients. The proposed criteria were derived to be protective of chl-a and DO concentrations. A reduction in nutrients was applied to meet the DO water quality target. The values under mechanistic modeling below represent the 90th percentile annual geometric mean nutrient concentrations from the nutrient reduction scenario.

A summary of proposed numeric nutrient criteria for TN, TP, and chl-a in Pensacola Bay segments are summarized in Table 2-15.

Table 2-15. Proposed and candidate numeric nutrient criteria for Pensacola Bay segments

		Proposed Criteria			Mechanistic Modeling		
SEGMENT	SEGMENT ID	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
Blackwater Bay	0201	0.53	0.022	3.9	0.53	0.022	3.9
Upper Escambia Bay	0202	0.43	0.025	3.7	0.43	0.025	3.7
East Bay	0203	0.50	0.021	4.2	0.50	0.021	4.2
Santa Rosa Sound	0204	0.34	0.018	4.1	0.34	0.018	4.1
Lower Escambia Bay	0205	0.44	0.023	4.0	0.44	0.023	4.0
Upper Pensacola Bay	0206	0.40	0.021	3.9	0.40	0.021	3.9
Lower Pensacola Bay	0207	0.34	0.020	3.6	0.34	0.020	3.6
Santa Rosa Sound	0208	0.33	0.020	3.9	0.33	0.020	3.9
Santa Rosa Sound	0209	0.36	0.020	4.9	0.36	0.020	4.9

2.2.8. Downstream Protective Values

In Pensacola Bay mechanistic models were applied to derive the proposed DPVs for TN and TP shown in Table 2-16.

Table 2-16. Proposed DPVs for Pensacola Bay

Tributary	LSPC Model Watershed ID	USGS Reach Code	Estuary Segment	TN (mg/L)	TP (mg/L)
Yellow River	20001	03140103001627	0201	0.51	0.021
Escambia River	20002	03140305001313	0202	0.43	0.027
Bayou Grande	20020	03140105000081	0207	1.23	0.012
Mulatto Bayou	20055	03140105000094	0205	1.98	0.102
East Bay River	20056	03140105000009	0203	0.73	0.015
Blackwater River	20057	03140104001202	0201	0.59	0.017
Bayou Texar	20058	03140105000051	0206	1.38	0.013
Trout Bayou	20059	03140105000096	0205	1.00	0.013
Dean Creek	20060	03140105000119	0203	0.78	0.006
Tom King Bayou	20076	03140105000149	0203	1.24	0.009
Williams Creek	20077	03140105000154	0209	0.97	0.014
Creek into Santa Rosa Sound	20078	03140105000079	0204	1.27	0.009

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2.3. Choctawhatchee Bay

2.3.1. Proposed Numeric Nutrient Criteria

Proposed numeric nutrient criteria for TN, TP, and chl-a in Choctawhatchee Bay segments are summarized in Table 2-17.

Table 2-17. Proposed numeric nutrient criteria for Choctawhatchee Bay segments

Segment Name	Segment Number	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
Eastern Choctawhatchee Bay	0301	0.47	0.025	8.1
Central Choctawhatchee Bay	0302	0.36	0.019	3.8
Western Choctawhatchee Bay	0303	0.21	0.012	2.4

2.3.2. General Characteristics

2.3.2.1. System Description

Choctawhatchee Bay is on the western Florida Panhandle and has three distinct areas: western, middle, and eastern (Ruth and Handley 2007).[14] Choctawhatchee Bay covers 129 mi^2 (334 km^2) and is about 27 mi (43 km) long (FDEP 2006). The watershed covers approximately 5,350 mi^2 (13,850 km^2) in Alabama and Florida, with approximately 42 percent of the watershed in Florida (FDEP 2011; Thorpe et al. 2002). Choctawhatchee River flows into the bay, and major tributaries into the Choctawhatchee River are Holmes, Wrights, Bruce, and Pine Log creeks. Alaqua, Rocky, Black, and Turkey creeks are direct tributaries of the bay. A portion of Sand Hill Lakes and a recharge area for Floridan aquifer springs, which discharge into Holmes Creek, are also included in the watershed. The East Pass (approximately 0.2 mi [0.32 km] in width), which is located to the immediate west of Destin, is the only direct outlet to the Gulf of Mexico (Thorpe et al. 2002).

The Choctawhatchee River is characterized by seasonal flooding and the interaction and mixing of freshwater and saltwater. This alluvial river is in a broad floodplain (FDEP 2006). Monthly rainfall averages over 30 years in the Choctawhatchee Bay area range from 3.2 to 9.4 in (8.1 to 23.9 cm). Summers typically receive more rainfall; fall and spring receive the least (NWFWMD 2011). Tropical storms and hurricanes are also typical of the area (NOAA No date). The average depth in the eastern portion of the bay is about 10 ft (3 m), while the western average depth is about 30 ft (9 m). The maximum depth is 43 ft (13 m) at Moreno Point (FDEP 2006).

Approximately 70 percent of the watershed is forested, forested wetlands, or clear-cut. Twenty percent of the watershed is agricultural, predominantly in the central portion of the watershed, and six percent consists of urban land uses scattered throughout the watershed and often

[14] The information presented in this system description was compiled to summarize local information pertaining to the proposed numeric nutrient criteria for this estuary. For more information on EPA's process of delineating, segmenting, and deriving numeric nutrient criteria for this estuary, please see Section 1.3.

clustered near small townships (Fry et al. 2011; NWFWMD 2009).[15] While urban land accounts for only a small portion of the overall land use of the watershed, development occurs mostly near the bay coastal areas. The northern portion of the bay is less developed because Eglin Air Force Base covers much of the land (Ruth and Handley 2007).

2.3.2.2. Impaired Waters[16]

There are 13 Class II and Class III marine WBIDs in the Choctawhatchee Bay area listed for a nutrient-related parameter on Florida's CWA section 303(d) list approved by EPA. Of the 13 WBIDs, five are Class II WBIDs and eight are Class III marine WBIDs. Of the five Class II WBIDs, one is impaired for nutrients (WBID 917) and four are impaired for DO (WBIDs 49A, 722, 679, and 986). All eight Class III marine WBIDs are impaired for DO (WBIDs 692, 843, 843B, 959, 959D, 959E, 959I, and 959J).[17]

One draft nutrient-related TMDL for Class II or Class III marine WBIDs exists in the Choctawhatchee Bay watershed, the *Boggy Bayou Nutrient and DO TMDL*, covering Class III marine WBID 692.[18]

2.3.2.3. Water Quality

An extensive literature search for Choctawhatchee Bay water quality data yielded very few studies. Most of the available water quality data are for the period before the 1990s. Surface waters are monitored by a volunteer water quality monitoring program, the Choctawhatchee Basin Alliance,[19] at 100 sites in the bay, including Choctawhatchee Bay, Choctawhatchee River, and Walton County's coastal dune lakes. Analyses are conducted for TP, TN, chl-a, water clarity, and color; surface and bottom measurements of temperature, oxygen, salinity, turbidity, and pH are taken. However, the collected data are not publicly available.

Based on a study from the mid-1970s, nitrogen concentrations levels decreased from east to west, with the lowest concentration at East Pass (Blaylock 1983). FDEP reported a long term geometric mean TN for the entire bay of 0.417 mg/L (416.92 µg/L)[20] between 1996 and 2009. During that time, concentrations were relatively stable until a spike in 2008 (FDEP 2010). FDEP analyzed Impaired Waters Rule (IWR) data for TP for the entire bay from 1996 to 2009, and

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[15] See Volume 1, Appendix C and its attachments for a summary of how land use data was combined and for a detailed breakdown of land uses.

[16] For more information about the data source, see Volume 1, Appendix A.

[17] The nutrient-related 303(d) list was a compilation of EPA-approved 303(d) listing information for nutrients, chl-a, and DO provided in three decision documents: September 2, 2009, Basin Groups 1, 2, and 5 EPA Decision Document (http://www.epa.gov/region4/water/tmdl/florida/documents/fl09303d_decisiondoc_090209.pdf);
May 13, 2010, Basin Group 3 EPA Decision Document (http://www.epa.gov/region4/water/tmdl/florida/documents/fl303d_%20partialapproval_decision_docs051410.pdf);
and the December 21, 2010, Basin Group 4 EPA Decision Document (http://www.epa.gov/region4/water/tmdl/florida/documents/group_4_final_dec_doc_and_partial_app_letter_12_21_10.pdf).

[18] TMDLs were identified in February 2011 by compiling nutrient-related draft/final TMDLs from the following three sources:
FDEP TMDL website (http://www.dep.state.fl.us/water/tmdl/index.htm)
EPA Region 4 website (http://www.epa.gov/region4/water/tmdl/florida/index.html)
EPA National WATERS expert query tool (http://www.epa.gov/waters/tmdl/expert_query.html)

[19] Choctawhatchee Basin Alliance water quality monitoring program: http://www.basinalliance.org/page.cfm?articleID=4

[20] Concentrations expressed in µg/L (in parentheses) were reported in µg/L in the literature.

found an initial increase from 1996 to 1998, a decrease between 1998 and 2002, and a stable period between 2002 and 2009. For the overall period, the long term geometric mean was 0.023 mg/L (22.7 µg/L)[21] (FDEP 2010). No further information was found in the available literature.

Stream runoff in the eastern portion of the bay reduces water clarity and limits the extent of seagrasses in the estuary (FFWCC 2011b). The freshwater inflow from the Choctawhatchee River also affects clarity in the bay. In a study published in the late 1980s, Secchi depth was lower in the eastern parts of the bay where the river enters the bay and higher in the western areas (Livingston 1987). FDEP reports that from the river delta to East Pass there is an east-to-west salinity gradient of 0 to 34 PSU (FDEP 2010). Blaylock's 1983 study found that factors affecting salinity in Choctawhatchee Bay include volume of river inflow, surface runoff, bathymetry of the system, and, to a lesser extent, tides (Blaylock 1983). Salinity levels in the bay were lowest from December through April when river flows peaked, and they were highest during the summer and fall months when flow was lowest (Livingston 1986). When freshwater inflows were lower, tidal saltwater exchange increased by forcing the more dense saltwater toward the river mouth (Blaylock 1983). Livingston (1987) found that bottom salinity levels increased from east to west in Choctawhatchee Bay. This pattern demonstrates the major influence that the Choctawhatchee River has on the main portions of the bay (Livingston 1987). Salinities in the eastern part of the bay, where the Choctawhatchee River meets the bay, were 2 to 5 PSU lower than the main bay. The western portion of the bay, closer to the Gulf of Mexico opening, had higher salinities than the eastern end closer to the Choctawhatchee River inflow (Blaylock 1983). Stratification in Choctawhatchee Bay creates a disruption in oxygen diffusion rates, which, combined with oxygen depletion, creates a lower DO layer in the bottom saline waters.

2.3.2.4. Biological Characteristics

Livingston (1986) noted that wetlands in the bay have been affected since the opening of East Pass in the late 1920s. The increased salinity from the Gulf of Mexico waters entering the bay, along with anthropogenic activities (urbanization, stormwater runoff, and sewage wastes), have also contributed to the changes (Livingston 1986). Using a Fish and Wildlife Research Institute (FWRI) GIS data layer from 1994 to 2006, FDEP estimated that 3.8 mi^2 (9.8 km^2 [2,411 ac]) of emergent marsh were in Choctawhatchee Bay (FDEP 2010).

Choctawhatchee Bay's freshwater inflow and poor flushing contribute to the extent of seagrass beds. Although seagrass coverage has fluctuated since the 1950s, it has generally been declining since the 1950s (Ruth and Handley 2007). Seagrass coverage measured in 1992 showed that 6.7 mi^2 (17 km^2 [4,261 ac]) of seagrasses were in Choctawhatchee Bay. Of those 6.7 mi^2 (17 km^2 [4,261 ac]), 83 percent (5.5 mi^2 [14 km^2 (3,536 ac)]) were in the western portion of the bay. By 2003, a 38 percent reduction of seagrass coverage had occurred, reducing coverage to 4.1 mi^2 (11 km^2 [2,623 ac]), and no seagrasses were observed in the eastern portion of the bay. Those losses in the eastern portion of the bay were attributed to increased CDOM from stream runoff, reduced water clarity, and fluctuating salinity. Monitoring in 2009 showed shoal grass (*Halodule wrightii*) to be the most prominent species of seagrass in the Choctawhatchee Bay, with turtle

[21] Concentrations expressed in µg/L (in parentheses) were reported in µg/L in the literature.

grass (*Thalassia testudinum*) in a few locations in the western end of the bay and near Santa Rosa Sound (FFWCC 2011b).

In 1985–1986, Livingston (1986) observed the highest annual average phytoplankton numbers in the western portions of Choctawhatchee Bay and at Old Pass Lagoon. Seasonally, peak numbers occurred in the bay during the spring and summer (Livingston 1986). Sampling in 1975 showed that in terms of biomass, diatoms were the dominant species in the bay from early spring to mid-summer, and that dinoflagellates would then become prevalent. In late fall and early winter, phytoplankton species were equally distributed between diatoms, dinoflagellates, and microalgae (Blaylock 1983). Zooplankton at the surface, which increased toward the west, also followed the east-west salinity gradient in the bay (Livingston 1986).

A series of red tide blooms affected Choctawhatchee Bay during 1999 and 2000, causing mass mortalities of fish, dolphins, and other wildlife (FDEP 2006).

Historically, the dominant infaunal invertebrates in Choctawhatchee Bay have been polychaete worms, tubificid worms, and other opportunistic species that are adapted to highly organic sediments and low DO. The key determinants of species distribution in the bay included salinity, depth, pollution, and seagrass beds (Livingston 1986). In general, high species richness was found in the western portion of Choctawhatchee Bay at seagrass-dominated sampling stations; lower species richness occurred in the middle of the bay (Livingston 1987).

The eastern oyster (*Crassostrea virginica*) is the primary harvested species of shellfish in Choctawhatchee Bay. The northern hard-shell clam, or quahog, and the brackish water or marsh clam are also present, although they are not found in sufficient quantities for commercial harvesting (FDEP 2010). Penaeid and periclimenid shrimp, blue crabs, and brief squid are also found in Choctawhatchee Bay (Livingston 1986).

In 1985–1986, Livingston (1986) found that the average annual numbers of epibenthic fish were highest near the river mouth, in various bayous, and in seagrass habitats in the western bay. Examples of dominant species found included spot (*Leiostomus xanthurus*), pinfish (*Lagodon rhomboides*), and bay anchovy (*Anchoa mitchilli*) (Livingston 1986). Major fisheries species included spotted seatrout (*Cynoscion nebulosus*), flounder (*Paralichthys albigutta*), king mackerel (*Scomberomorus cavalla*), and red snapper (*Lutjanus campechanus*), among others (FFWCC 2011a; Livingston 1986).

For a more detailed summary of this water body, see Volume 1, Appendix A.

2.3.3. Data Used

Several data sources specific to Choctawhatchee Bay were used in addition to those sources described in Section 1.4.3, as summarized in Table 2-18.

Table 2-18. Data sources specific to Choctawhatchee Bay models

Data	Source (citation)	Location Used
Municipal and industrial point sources	Alabama Department of Environmental Management (ADEM No date a, No date b)	Choctawhatchee watershed model
Water quality data	Alabama Department of Environmental Management (ADEM 2006)	Choctawhatchee watershed model
FDEP Level III Florida Land Use	Northwest Florida Water Management District (NWFWMD 2009)	Choctawhatchee watershed model

2.3.4. Segmentation

The isohaline GIS analysis yielded three segments for Choctawhatchee Bay: western, middle, and eastern. That is consistent with segmentation scheme used in the USGS seagrass assessment study (Ruth and Handley 2007). The three segments differ in the seagrass distribution. Figure 2-5 shows the resulting three segments for Choctawhatchee Bay.

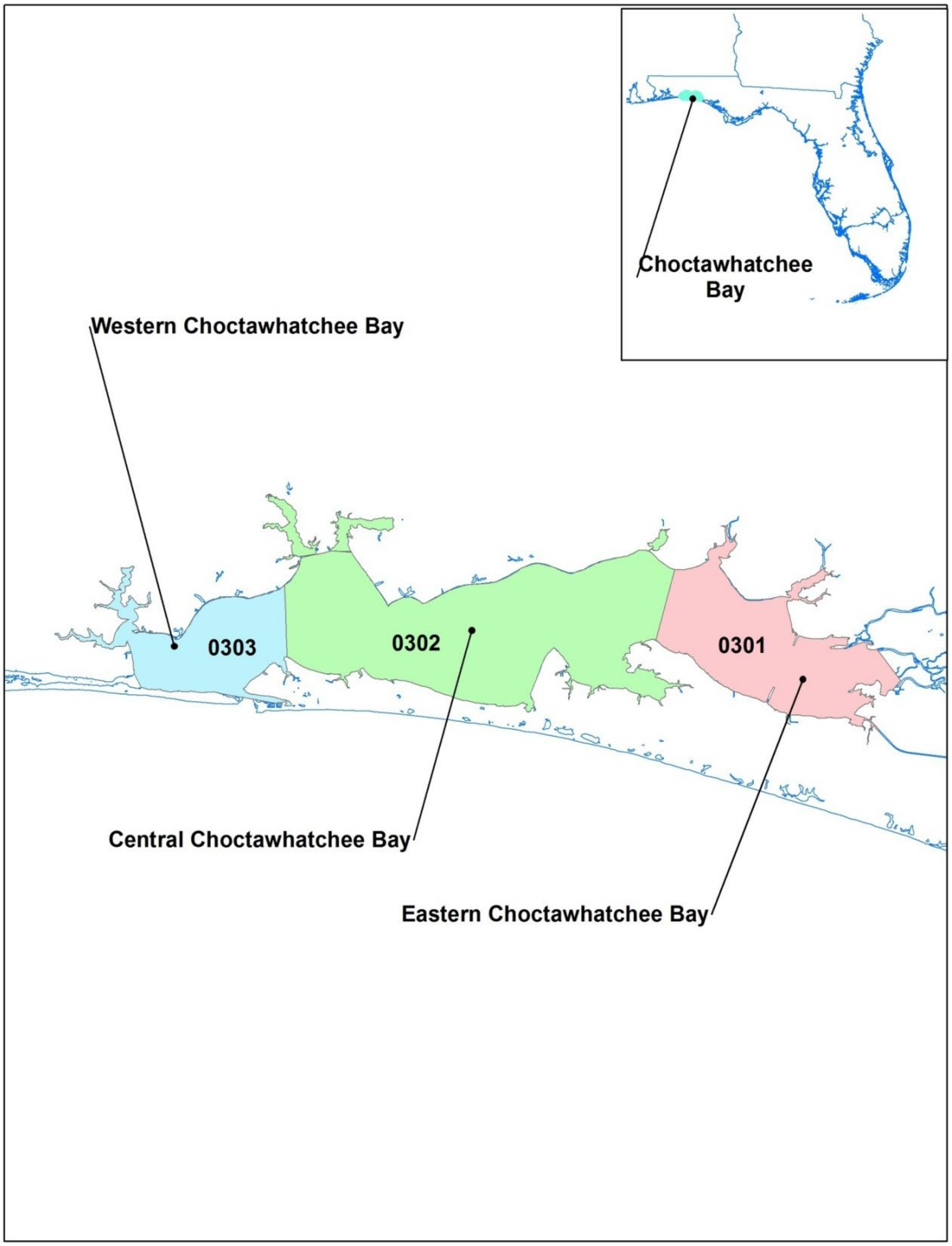

Figure 2-5. Results of Choctawhatchee Bay segmentation

2.3.5. Water Quality Targets

2.3.5.1. Seagrass Depth and Water Clarity Targets

Seagrass depth of colonization (measured as Z_c) and water clarity (measured as K_d) targets were established by analyzing the 1992 seagrass coverage (Figure 2-6) (USGS 2012). Seagrass depth of colonization and water clarity targets were computed for segments 0302 and 0303. There was no seagrass present in estuary segment 0301 (Table 2-19).

Figure 2-6. Map of 1992 Seagrass coverage in Choctawhatchee Bay. Seagrass is indicated as continuous (green) or discontinuous (teal). Estuary segmentation scheme is indicated in grey. Irregular north-south lines in central and eastern Choctawhatchee Bay are bridges.

Table 2-19. Choctawhatchee Bay seagrass depth and water clarity targets by segment

Estuary Segment	Depth of Colonization (Z_c) Target (m)	Light Attenuation Coefficient (K_d) Target (1/m)
0301	No target	-
0302	2.7	0.6
0303	3.3	0.5

2.3.5.2. Chlorophyll a *Target*

To prevent nuisance algal blooms and protect the estuary's designated uses, chl-a levels must not exceed 20 µg/L more than 10 percent of the time. The rationale for that target is provided in Section 1.2.2.

2.3.5.3. Dissolved Oxygen Targets

On the basis of the rationale that sufficient DO is necessary to protect aquatic life, as described in Section 1.2.3, the following DO targets were established:

- Minimum allowable DO of 4.0 mg/L as a water column average for each estuary segment 90 percent of the time over the simulation's time span
- Daily average DO of 5.0 mg/L as a water column average for each estuary segment 90 percent of the time over the simulation's time span

- Minimum 3-hour average DO of 1.5 mg/L in the bottom two layers for each estuary segment over the simulation's time span

2.3.6. Results of Analyses

2.3.6.1. Mechanistic Model Analysis

Average load contributions from the Choctawhatchee watershed are shown in Table 2-20.

Table 2-20. Average load contributions from the Choctawhatchee watershed (2002–2009)

	TN Load[1] (kg/d; Mean ± se)			TP Load[1] (kg/d; Mean ± se)		
Year	Existing[2]	Background[3]	Anthropogenic[4]	Existing[2]	Background[3]	Anthropogenic[4]
2002	9657 ± 523	5,462 ± 357	4,195 ± 205	653 ± 40	374 ± 35	279 ± 8
2003	21,023 ± 1,138	12,198 ± 829	8,825 ± 369	1,126 ± 69	720 ± 61	406 ± 11
2004	14,966 ± 906	8,205 ± 599	6,761 ± 336	944 ± 68	551 ± 58	393 ± 12
2005	17,822 ± 1,085	9,530 ± 650	8,292 ± 474	1,077 ± 65	591 ± 51	486 ± 16
2006	6,021 ± 271	2,889 ± 142	3,132 ± 151	694 ± 24	212 ± 15	481 ± 16
2007	7,040 ± 322	3,329 ± 164	3,712 ± 184	684 ± 25	245 ± 17	438 ± 11
2008	12,464 ± 608	6,127 ± 331	6,337 ± 319	864 ± 40	362 ± 28	502 ± 15
2009	23,659 ± 2,006	13,409 ± 1,307	10,250 ± 754	1,509 ± 126	884 ± 108	624 ± 22

[1] Annual average daily load from all terminal reaches
[2] Annual average daily load from all terminal reaches computed from existing model scenario
[3] Annual average daily load from all terminal reaches computed from background model scenario (remove anthropogenic sources/land use; use existing hydrology)
[4] Anthropogenic load computed as difference between existing and background load

For Choctawhatchee Bay, the chl-a target was met for each segment according to 2002–2009 nutrient loads. The DO targets were met according to 2002–2009 nutrient loads, except for the daily water column average target of 5 mg/L, which could not be met for any segment using either the 2002–2009 nutrient loads scenario or the non-anthropogenic nutrient scenario. The light attenuation coefficient based on the depth of colonization target could not be met for segments 0302 and 0303 using either the light attenuation coefficient based on 2002–2009 nutrient loads or the light attenuation coefficient based on the non-anthropogenic nutrient scenario. Table 2-21 identifies which targets were met under 2002–2009 nutrient loads and which targets were not met.

An evaluation of model sensitivity to the water quality targets applied revealed that the DO and light targets were insensitive to changes in nutrients in Choctawhatchee Bay.

Table 2-21. Water quality targets met for Choctawhatchee Bay based on mechanistic modeling

Segment	DO	Chl-a	K_d
0301	No	Yes	No target
0302	No	Yes	No
0303	No	yes	No

A summary of candidate criteria for Choctawhatchee Bay Estuary segments based 2002–2009 nutrient loads is given in Table 2-22.

Table 2-22. Summary of candidate criteria for Choctawhatchee Bay derived from mechanistic modeling

Segment	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
0301	0.61	0.035	4.1
0302	0.48	0.017	3.9
0303	0.41	0.015	3.5

2.3.6.2. Statistical Model Analysis

Annual geometric mean light attenuation coefficient increased strongly with increased chl-a in segment 0302, and also exhibited an increasing relationship in segment 0303 (Figure 2-7). In segment 0302, chl-a concentrations associated with meeting the light attenuation coefficient target were within the range of data, but in segment 0303, the predicted chl-a concentration for meeting the light attenuation coefficient target was well below the lower bound of the observed data, so the candidate chl-a value associated with the water clarity endpoint for segment 0303 is based on the lower bound of the data used to estimate the relationship.

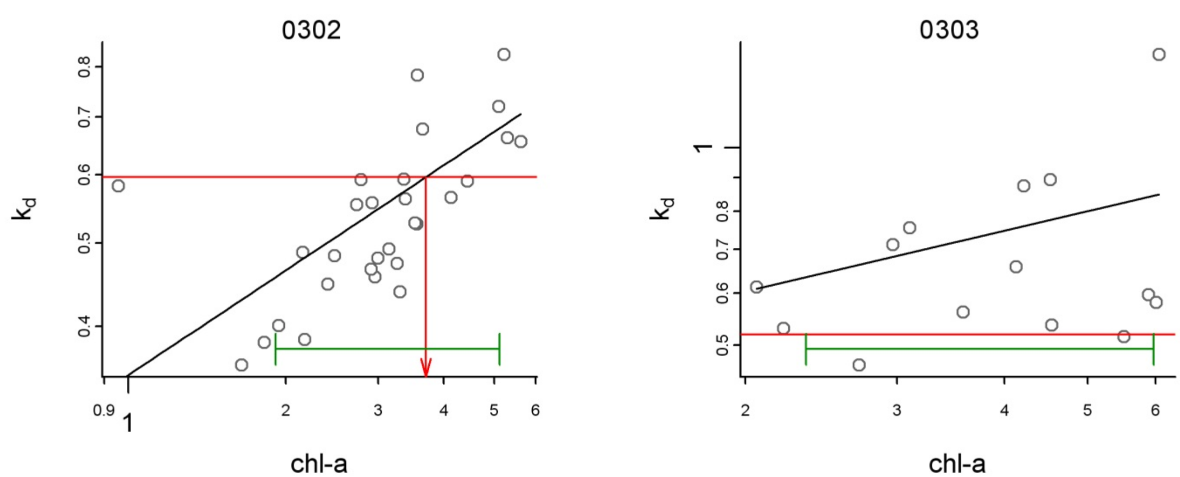

Figure 2-7. Relationships between annual geometric light attenuation coefficient (K_d) and chl-a in Choctawhatchee Bay. Solid black line: segment-wide relationship; red horizontal line: K_d target; red vertical arrow: chl-a concentrations associated with K_d target; green line segment: 5th to 95th percentile range of chl-a concentrations, open circles: observed annual geometric mean K_d and chl-a concentrations.

Estimates of chl-a concentrations associated with a phytoplankton bloom frequency of 10 percent were generally higher than the upper bound of the range of chl-a concentrations observed in all segments of Choctawhatchee Bay (Figure 2-8). Based on the available data, EPA did not find evidence to support the use of the bloom frequency endpoint in segments 0302 and 0303. Hence, candidate chl-a criteria associated with the phytoplankton bloom endpoint were based on the upper bound of the data for all segments. Based on these analyses, chl-a criteria were derived that protected both the water clarity and phytoplankton bloom endpoints (Table 2-23).

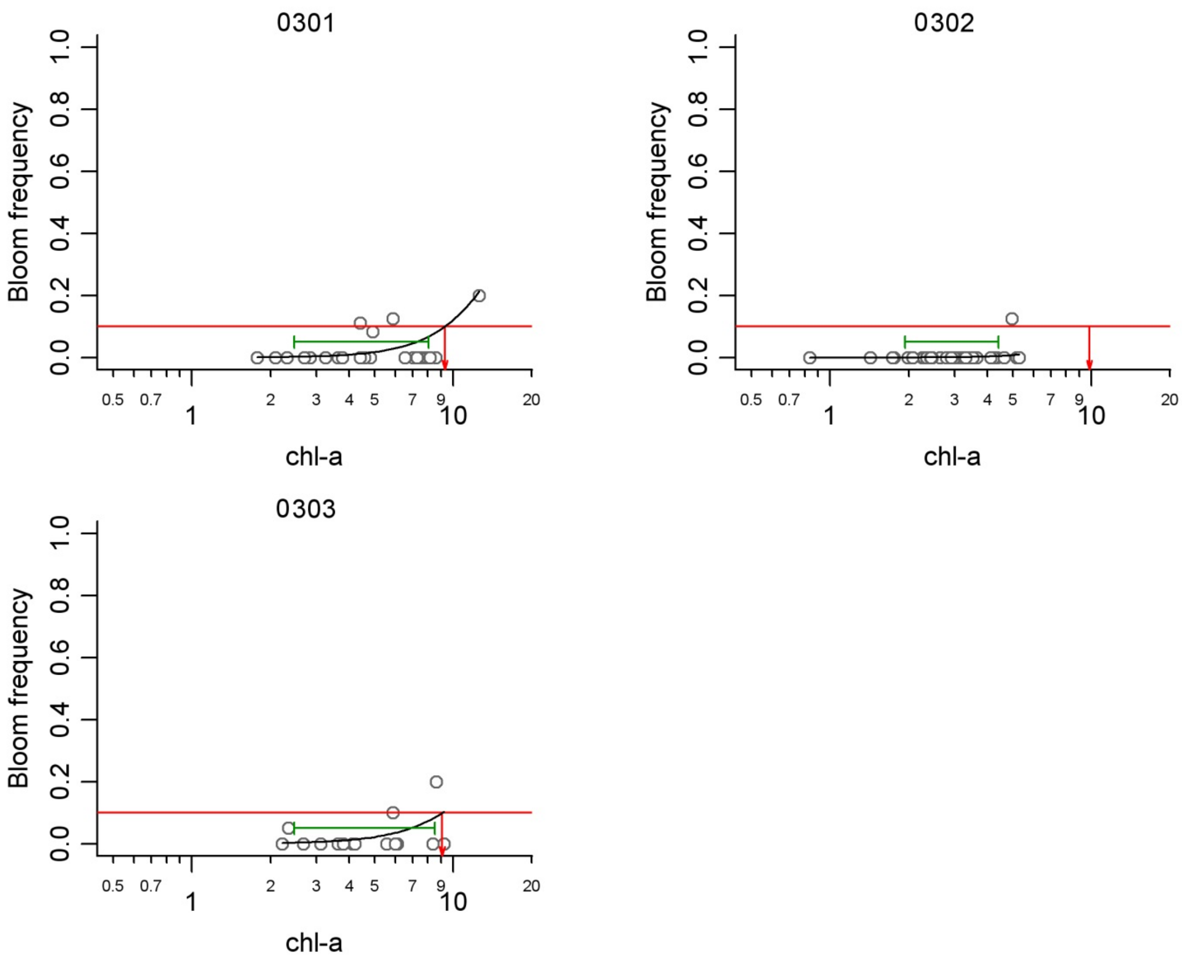

Figure 2-8. Estimates of annual geometric chl-a concentrations associated with bloom frequency of 0.1 in Choctawhatchee Bay. Red horizontal line: bloom frequency of 0.1, red vertical arrow: annual geometric mean chl-a concentration associated with 0.1 bloom frequency, green line segment: 5th to 95th percentile range of observed data.

Table 2-23. Summary of candidate chl-a criteria. No seagrass present in segment 0301, so no chl-a criteria associated with clarity was calculated. Values with asterisks indicate that the predicted candidate criterion was greater than the upper bound of chl-a values, or less than the lower bound of chl-a values used in estimating the empirical relationship, so listed criterion is based on the upper or lower bound of the data.

Segment	Chl-a (clarity) (µg/L)	Chl-a (bloom) (µg/L)	Chl-a (final) (µg/L)
0301	-	8.1*	8.1
0302	3.8	4.4*	3.8
0303	2.4*	8.5*	2.4

Relationships between annual geometric mean TP, TN, and chl-a were estimated and used to derive TN and TP criteria associated with candidate chl-a criteria (Figure 2-9 and Figure 2-10). Chl-a concentrations increased with increasing TP and TN in all segments, although relationships were comparatively weaker in segments 0301 and 0303 compared to segment 0302. This difference in nutrient-chl-a relationships among segments seems reasonable given the strong influence of marine waters in segment 0303, and the likely high concentrations of suspended sediment and CDOM in segment 0301 near the tributary mouth. Substantial differences in the relationships were also observed among stations within each segment (see the difference in the grey lines), but, as with differences among segments, difference among stations is expected, given differences in station locations and differences in the natural physical and chemical characteristics at these stations.

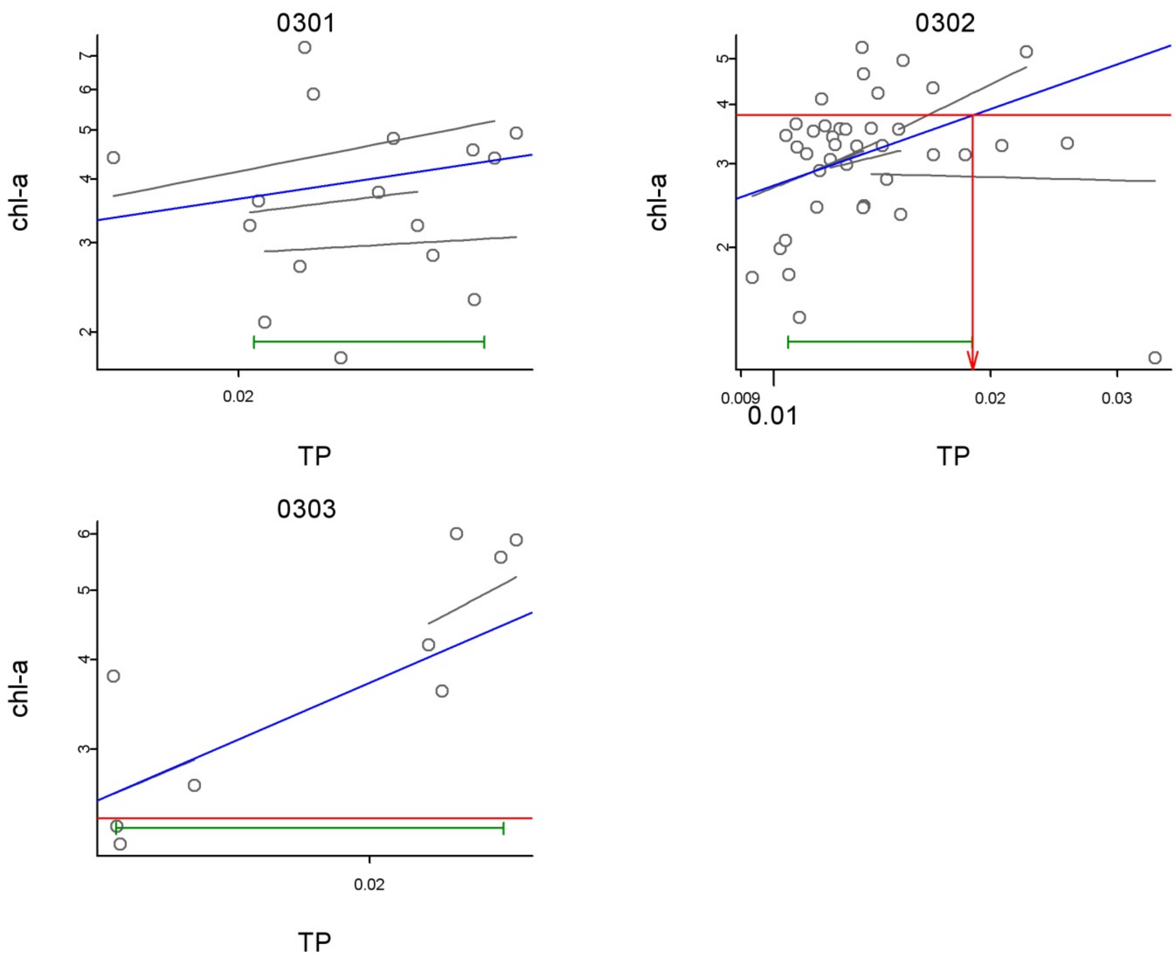

Figure 2-9. Relationships between TP and chl-a in Choctawhatchee Bay. Open circles: observed annual average values of TP and chl-a, red horizontal line: chl-a criterion, red vertical arrow: TP criterion associated with chl-a criterion, blue line: estimated segment-wide relationship between TP and chl-a, grey lines: estimated station-specific relationships between TP and chl-a, green line segment: 5th to 95th percentile range of observed annual geometric mean TP concentrations.

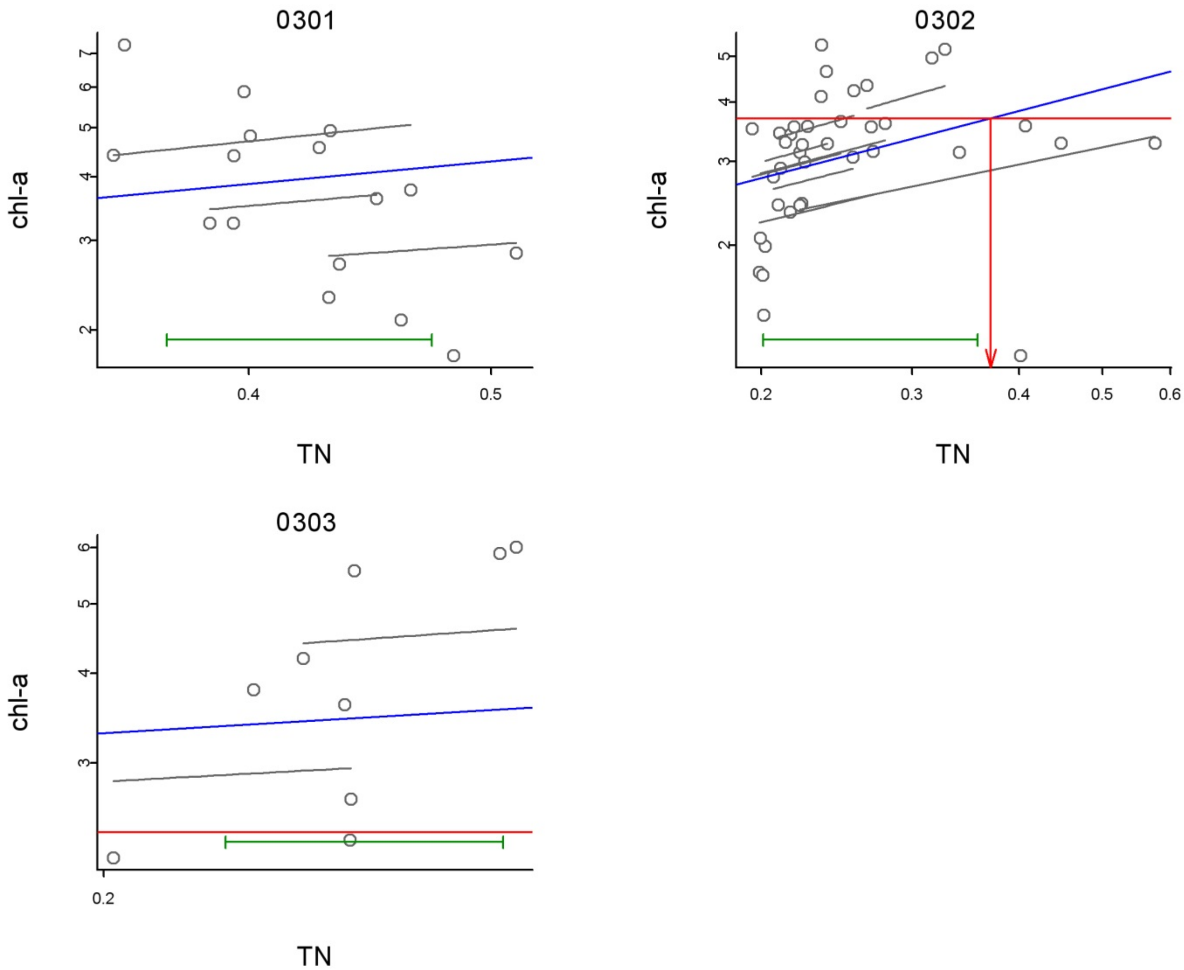

Figure 2-10. Relationships between TN and chl-a in Choctawhatchee Bay. Open circles: observed annual average values of TN and chl-a, red horizontal line: chl-a criterion, red vertical arrow: TN criterion associated with chl-a criterion, blue line: estimated segment-wide relationship between TN and chl-a, grey lines: estimated station-specific relationships between TN and chl-a, green line segment: 5th to 95th percentile range of observed annual geometric mean TN concentrations.

In segment 0301, observed chl-a concentrations were well below the derived candidate chl-a criterion, and increased chl-a concentrations were associated with increases in both TN and TP. Hence, TN and TP criteria for this segment were based on the upper bound of observed values (Table 2-24). In segments 0302 and 0303, derived TP criteria values associated with the candidate chl-a criterion were within the range of observed values, and these values are the proposed criteria. In segment 0302, the derived TN criterion was slightly higher than the upper bound of observed data, so the proposed criterion is based on the upper bound of observed values. Conversely, in segment 0303, the derived TN criterion was lower than the lower bound of the observed values, so the proposed criterion is based on this lower bound.

Table 2-24. Summary of candidate TN and TP criteria in Choctawhatchee Bay. Values with asterisks indicate that the predicted candidate criterion was greater than the upper bound of observed values, or less than the lower bound of observed values used in estimating the empirical relationship, so listed criterion is based on the upper or lower bound of the data.

Segment	TN (mg/L)	TP (mg/L)
0301	0.47*	0.025*
0302	0.36*	0.019*
0303	0.21*	0.012*

2.3.7. Application of Analyses for Proposed Numeric Nutrient Criteria

Data were sufficient to use statistical analyses as the primary line of evidence when deriving criteria. Relationships between light attenuation coefficient and chl-a, between chl-a and phytoplankton bloom frequency, and among TN, TP, and chl-a all conformed with expectations, and hence, these relationships were used to derive criteria. In some segments and for some parameters the predicted criteria values were not within the observed range of data, and in these cases, criteria values were based on the upper or lower bound of the observed range of the nutrient variable.

EPA also evaluated the sensitivity of three endpoints in the mechanistic modeling approach for Choctawhatchee Bay: (1) water clarity protective of historic depth of seagrasses, (2) chl-a concentrations associated with balanced phytoplankton biomass, and (3) DO concentrations sufficient to maintain aquatic life. EPA found the water clarity and DO endpoints were not met in all segments under the calibrated 2002–2009 nutrient loads and were insensitive to changes in nutrients loads. As a result, the water clarity and DO endpoints were not used in Choctawhatchee Bay. The chl-a target was met and was shown to be sensitive to changes in nutrients. The proposed criteria were derived to be protective of chl-a concentrations. The values under mechanistic modeling represent the 90th percentile annual geometric mean nutrient concentrations from the 2002–2009 modeled nutrient loads.

Proposed numeric nutrient criteria for TN, TP, and chl-a in Choctawhatchee Bay segments are summarized in Table 2-25. Criteria values from mechanistic modeling were generally comparable to corroborated values derived from statistical analysis, given inherent uncertainties in both the mechanistic and statistical models.

Table 2-25. Proposed and candidate numeric nutrient criteria for Choctawhatchee Bay segments

		Proposed Criteria			Mechanistic Modeling			Statistical Modeling		
SEGMENT	SEGMENT ID	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
Eastern Choctawhatchee Bay	0301	0.47	0.025	8.1	0.61	0.035	4.1	0.47	0.025	8.1
Central Choctawhatchee Bay	0302	0.36	0.019	3.8	0.48	0.017	3.9	0.36	0.019	3.8
Western Choctawhatchee Bay	0303	0.21	0.012	2.4	0.41	0.015	3.5	0.21	0.012	2.4

2.3.8. Downstream Protective Values

In Choctawhatchee Bay mechanistic models were applied to derive proposed DPVs for TN and TP shown in Table 2-26.

Table 2-26. Proposed DPVs for Choctawhatchee Bay

Tributary[a]	LSPC Model Watershed ID	USGS Reach Code	Estuary Segment	TN (mg/L)	TP (mg/L)
Fourmile Creek	30002	03140102000034	0301	0.98	0.013
Bear Creek	30003	03140102000440	0301	0.87	0.011
Alaqua Creek	30008	03140102001555	0301	0.58	0.008
Basin Creek	30009	03140102000692	0302	0.68	0.005
Mullet Creek	30010	03140102000338	0302	0.86	0.006
	30011	03140102000713	0302	1.15	0.007
	30012	03140102000874	0302	1.43	0.008
Rocky Creek	30017	03140102002756	0302	0.67	0.007
Toms Creek	30021	03140102000273	0302	0.69	0.008
Lightwood Knot Creek	30022	03140102000259	0303	0.64	0.007
	30023	03140102000264	0303	1.29	0.009
Black Creek	30174	03140203002447	0301	0.72	0.013
Choctawhatchee River	30176	03140203001448	0301	0.57	0.078
	30225	03140102001646	0303	1.72	0.013
Turkey Creek	30226	03140102000153	0302	0.54	0.006

[a] Tributary names left blank are unnamed

2.3.9. References

ADEM. 2006. *FY06 Master Water Quality Database*. Alabama Department of Environmental Management. Provided by ADEM to the United States Environmental Protection Agency.

ADEM. No date a. *Alabama Municipal Point Source Coverage*. Alabama Department of Environmental Management. Provided by ADEM to the United States Environmental Protection Agency.

ADEM. No date b. *Alabama Industrial Point Source Coverage*. Alabama Department of Environmental Management. Provided by ADEM to the United States Environmental Protection Agency.

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2.4. St. Andrews Bay

2.4.1. Proposed Numeric Nutrient Criteria

Proposed numeric nutrient criteria for TN, TP, and chl-a in St. Andrews Bay segments are summarized in Table 2-27.

Table 2-27. Proposed numeric nutrient criteria for St. Andrews Bay segments

Segment Name	Segment Number	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
East Bay	0401	0.31	0.014	4.6
St. Andrews Sound	0402	0.14	0.009	2.3
Eastern St. Andrews Bay	0403	0.24	0.021	3.9
Western St. Andrews Bay	0404	0.19	0.016	3.1
Southern St. Andrews Bay	0405	0.15	0.013	2.6
North Bay 1	0406	0.22	0.012	3.7
North Bay 2	0407	0.22	0.014	3.7
North Bay 3	0408	0.21	0.016	3.4
West Bay	0409	0.23	0.022	3.8

2.4.2. General Characteristics

2.4.2.1. System Description

St. Andrews Bay[22] is near the middle of the Florida Panhandle. It is commonly divided into four hydrologically linked bays (West Bay, North Bay, East Bay, and St. Andrews Bay proper) [23] (FDEP 2010). St. Andrews Sound is an adjacent, unattached lagoon system (Brim and Handley 2007). The entire system is about 94 mi^2 (243 km^2). West Pass is a man-made, maintained access way to the Gulf of Mexico in the southern portion of St. Andrews Bay proper. East Pass is an intermittent passage to the Gulf just northwest of St. Andrews Sound. Only minor sources of freshwater feed the St. Andrews Bay system, with Econfina Creek, Deer Point Lake, the Gulf Intracoastal Waterway (GIWW), and smaller creeks and streams as the primary inflow of freshwater (FDEP 2010). Because no major river discharges into the bay, there is little sedimentation and associated turbidity, and salinities remain high. The bay is relatively deep

[22] St. Andrews Bay is also referred to as St. Andrew Bay.

[23] The information presented in this system description was compiled to summarize local information pertaining to the proposed numeric nutrient criteria for this estuary. For more information on EPA's process of delineating, segmenting, and deriving numeric nutrient criteria for this estuary, please see Section 1.3.

with an average depth of 17–27 ft (5–8 m) (Keppner and Keppner 2001) and a maximum depth of 40 ft (12 m) (Brim and Handley 2007). Most of the watershed is in Bay County, and Panama City is the largest municipality surrounding the bay (Brim and Handley 2007). St. Andrews State Park Aquatic Preserve, St. Andrews State Recreation Area, Lake Powell, Phillips Inlet, and all tributaries are designated as OFWs.[24]

Tides enter St. Andrews Bay through the West Pass and spread from St. Andrews Bay proper to East, West, and North bays. The tide primarily shows diurnal characteristics and has an amplitude of about 6 in (15 cm) (Blumberg and Kim 2000). Tidal vertical amplitudes can range from 0.2 to 2.2 ft (0.06 to 0.67 m) (Brim and Handley 2007). Average flows are driven by low-frequency currents, which are driven by winds and gulf water level fluctuations. Average water flow out of St. Andrews Bay into the Gulf of Mexico was around 6,400 cfs (180 m^3/s) between April 1994 and January 1995. Over the same period the inflow from East Bay was 3,500 cfs (100 m^3/s), and 1,400 cfs (40 m^3/s) through West Bay. When low frequency flow rates taken from 1994–1995 data were averaged and put into a model of St. Andrews Bay, flow was found to have two distinct levels, with different fluxes in the surface and bottom layers. For example, in West Bay the surface layer flow is greater than the bottom layer, resulting in a net flow toward the east (Blumberg and Kim 2000). Predominate land uses in the St. Andrews Bay watershed are forested or clear-cut (50%), forested wetland (30%), urban (13%), and agriculture (4%). The urban area is concentrated in the central coastal region of the watershed (Fry et al. 2011; NWFWMD 2009).[25]

2.4.2.2. Impaired Waters[26]

There are 15 Class II and Class III marine WBIDs in the St. Andrews Bay area listed for a nutrient-related parameter on Florida's CWA section 303(d) list approved by EPA. Of the 15 WBIDs, eight are Class II WBIDs and seven are Class III marine WBIDs. Of the eight Class II WBIDs, one is impaired for DO (WBID 1142), one is impaired for nutrients (WBID 1110), four are impaired for nutrients and DO (WBIDs 1088, 1123, 1128, and 1131), and two are impaired for nutrients, chl-a, and DO (WBIDs 1053A and 1141B). Of the seven Class III marine WBIDs, three are impaired for DO (WBIDs 1009A, 1037, and 1055A), two are impaired for nutrients (WBIDs 1136 and 1170), and two are impaired for nutrients and DO (WBIDs 1027A and 1144).[27]

[24] Section 62-302.700, F.A.C.

[25] See Volume 1, Appendix C and its attachments for a summary of how land use data was combined and for a detailed breakdown of land uses.

[26] For more information about the data source, see Volume 1, Appendix A.

[27] The nutrient-related 303(d) list was a compilation of EPA-approved 303(d) listing information for nutrients, chl-a, and DO provided in three decision documents: September 2, 2009, Basin Groups 1, 2, and 5 EPA Decision Document (http://www.epa.gov/region4/water/tmdl/florida/documents/fl09303d_decisiondoc_090209.pdf);
May 13, 2010, Basin Group 3 EPA Decision Document
(http://www.epa.gov/region4/water/tmdl/florida/documents/fl303d_%20partialapproval_decision_docs051410.pdf);
and the December 21, 2010, Basin Group 4 EPA Decision Document
(http://www.epa.gov/region4/water/tmdl/florida/documents/group_4_final_dec_doc_and_partial_app_letter_12_21_10.pdf).

One final nutrient-related TMDL for Class II or Class III marine WBIDs exists in the St. Andrews Bay watershed, the *St. Andrew Bay Nutrient and DO TMDL*. It covers Class II and III marine WBIDs 1088, 1123, 1128, 1131, 1141B, 1144, and 1172.[28]

2.4.2.3. Water Quality

The 95th percentile of DO values measured between 1990 and 2006 ranged from 9.8 mg/L for surface waters to 9.1 mg/L for bottom waters (Hemming et al. 2011). FDEP analyzed DO measurements collected from 1967 to 2010, separating results into recent (2000–2010) and historic (1967–1999) periods. Mean DO levels were similar in recent (6.34 to 7.24 mg/L) and historic (6.58 to 7.39 mg/L) periods (FDEP 2010).

According to measurements from 1990 to 2006, the 95th percentile turbidity values for all surface and bottom waters were 5.2 and 8.3 nephelometric turbidity units (NTU), respectively, and the overall Secchi depth was 15.0 ft (4.6 m) (Hemming et al. 2011). Salinity in St. Andrews Bay is usually higher in St. Andrews Bay proper, closer to the Gulf of Mexico passes, compared to the outer bays. A report published in 2001 found that measurements in the main bay consistently exceeded 30 PSU annually. The East, West, and North bays (farther north and farther away from the Gulf Passes) have reached salinity levels as low as 10 PSU after heavy rain. Stratification occurs throughout the deeper portions of the bays, with the colder, more saline waters at bottom and mid-depths (Keppner and Keppner 2001).

Annual geometric mean chl-a concentrations ranged from 1.9 to 4.5 µg/L during 2000–2009 sampling for the IWR. After 2003 a decreasing trend was noted, and the long-term geometric mean was 2.8 µg/L. Of the samples collected, approximately one-third had a chl-a level below the minimum detect limit of 5 µg/L (FDEP 2010). The 95th percentile for chl-a values was 11 µg/L between 1990 and 2006 (Hemming et al. 2011). IWR data collected from January 2000 to December 2009 showed that TN levels fluctuated (but were low), and annual geometric mean levels ranged from 0.277 to 0.561 mg/L, with a long-term geometric mean of 0.403 mg/L (FDEP 2010). IWR data also showed the annual geometric mean for TP to be low, ranging from 0.012 to 0.025 mg/L, with a long-term geometric mean of 0.016 mg/L (FDEP 2010). The 95th percentile for 1990–2006 University of Florida data were 0.610 mg/L for TN and 0.041 mg/L for TP (Hemming et al. 2011).

2.4.2.4. Biological Characteristics

More than 6,300 documented plant and animal species live in the St. Andrews Bay system. More than 2,900 species, including more than 300 fish species, occur in the bay (Friends of St. Andrew Bay 2009). Salt marsh and inland forested wetlands are primarily in the West Bay area but occur throughout the watershed. Salt marshes in the bay contain extensive stands of needle rush (*Juncus roemerianus*), found at, above, and below the mean high tidal line. The intertidal zone is dominated by smooth cordgrass (*Spartina alterniflora*) (Thorpe et al. 2000).

[28] TMDLs were identified in February 2011 by compiling nutrient-related draft/final TMDLs from the following three sources:
FDEP TMDL website (http://www.dep.state.fl.us/water/tmdl/index.htm)
EPA Region 4 website (http://www.epa.gov/region4/water/tmdl/florida/index.html)
EPA National WATERS expert query tool (http://www.epa.gov/waters/tmdl/expert_query.html)

Turtle grass (*Thalassia testudinum*) is the dominant seagrass in the St. Andrews Bay system. It grows at depths of 6–8 ft (1.8–2.4 m) in areas near the gulf (St. Andrews Bay proper and St. Andrews Sound); growth is limited by light in interior segments (maximum growth depths of 4–6 ft [1.2–1.8 m]). In shallow and intertidal bay areas, shoal grass (*Halodule wrightii*) dominates. Interspersed throughout the turtle grass are patches of manatee grass (*Syringodium filiforme*), with some pure stands near clear, highly saline waters entering the bay from the gulf. Widgeon grass (*Ruppia maritima*) has been observed in less saline areas of the bay; rare star grass (*Heteranthera zosterifolia*) and paddle grass (*Halophila* spp.) have also been observed in the southern portion of the bay (Brim and Handley 2007).

In 1992 there were 6.6 mi^2 (17 km^2 [4,225 ac]) of continuous seagrass beds and 8.8 mi^2 (23 km^2 [5,607 ac]) of patchy beds in the St. Andrews Bay system. That was an 8 percent (3.1 mi^2 [8.1 km^2 (2,011 ac)]) decrease from seagrass coverage in 1953. In the St. Andrews Bay proper segment alone, about 0.39 mi^2 (1.0 km^2 [250 ac]) were lost from 1964 to 1992 (Brim and Handley 2007). From 1992 to 2003 coverage in the whole bay system increased 14 percent to 18 mi^2 (45 km^2 [11,232 ac]) (FFWCC 2011). East Bay seagrass increased, with 1.0 mi^2 (2.6 km^2 [650 ac]) of continuous coverage in 1953 to 2.5 mi^2 (6.6 km^2 [1,631 ac]) in 1992. North Bay lost about 0.21 mi^2 (0.54 km^2 [134 ac]) from 1953 to 1992. The majority of the loss occurred on the south and southwest shoreline. West Bay experienced the most loss, dropping 82 percent of continuous grass beds from 1980 to 1992 (from 2.1 mi^2 to 0.36 mi^2 [5.4 km^2 (1,343 ac) to 0.92 km^2 (227 ac)]), and losing 43 percent of patchy beds from 1953 to 1992 (from 4.7 mi^2 to 2.7 mi^2 [12 km^2 (3,037 ac) to 7.0 km^2 (1,725 ac)]) (Brim and Handley 2007). From 1992 to 2003, seagrass coverage in West Bay increased 30 percent (0.91 mi^2 [2.4 km^2 (585 ac)]) (FFWCC 2011).

East Bay, West Bay, and North Bay and a portion of the tributaries to these bays are Class II waters, suitable for shellfish harvesting.[29] Those waters support commercial oyster and commercial and recreational bay scallop (*Argopecten irradians*) fisheries (FDEP 2010; Thorpe et al. 2000). Restoration efforts were implemented after a decline in bay scallop population was found in 2008 (Arnold et al. 2009). NOAA's 1992 distribution and abundance report summarizes species found in Gulf of Mexico estuaries. Blue crab was the only species in St. Andrews Bay that was highly abundant (numerically dominant relative to other species). Abundant species (often encountered in substantial numbers relative to other species) include bay squid, brown shrimp, pink shrimp, and grass shrimp. Bay scallop, American oyster, common rangia, hard clam, white shrimp, and spiny lobsters were common species (species frequently encountered but not in large numbers; presence does not imply a uniform distribution over a specific salinity zone); gulf stone crab was rare (definitely present but not frequently encountered); and stone crab was not present (Nelson et al. 1992).

Oyster reefs are the only natural hardbottom community in the bay; rock jetties, cement pilings, and other artificial substrates are the primary hardbottom habitats. Those with numerous cracks/crevasses support a wide variety of communities, from sessile and motile organisms to economically important finfish (Thorpe et al. 2000). Many recreationally important fish occur, including striped mullet (*Mugil cephalus*), Spanish mackerel (*Scomberomorus maculates*), and

[29] Section 62-302.400, F.A.C.

bluefish (*Pomatomus saltatrix*) (Bergquist et al. 1997). Highly abundant species in NOAA's 1992 distribution and abundance survey include gulf menhaden (*Brevoortia patronus*), hardhead catfish (*Arius felis*), pinfish (*L. rhomboides*), and striped mullet (*M. cephalus*). Some of the abundant species include Spanish mackerel (*S. maculates*), gulf flounder (*Paralichthys albigutta*), spotted seatrout (*Cynoscion nebulosus*), silver perch (*Bairdiella chrysoura*), bluefish (*P. saltatrix*), and bay anchovy (*Anchoa mitchilli*) (Nelson et al. 1992).

For a more detailed summary of this water body, see Volume 1, Appendix A.

2.4.3. Data Used

One data source specific to St. Andrews Bay was used in addition to those sources common to all estuary systems (described in Section 1.4.3), as summarized in Table 2-28.

Table 2-28. Data source specific to St. Andrews Bay models

Data	Source	Location Used
Springs discharge and water quality data	Florida Department of Environmental Protection (FDEP 2004, 2009)	St. Andrews watershed model

2.4.4. Segmentation

The isohaline GIS analysis and geomorphological considerations yielded five segments in St. Andrews Bay: St. Andrews Bay proper, East Bay, North Bay, West Bay, and St. Andrews Sound (a separate lagoonal embayment to the southeast). That is consistent with the segmentation scheme used in the USGS seagrass assessment study (Brim and Handley 2007). St. Andrews Bay proper and North Bay were further subdivided into three segments apiece to account for the variation in seagrass depth distribution. The resulting nine segments for St. Andrews Bay are shown in Figure 2-11.

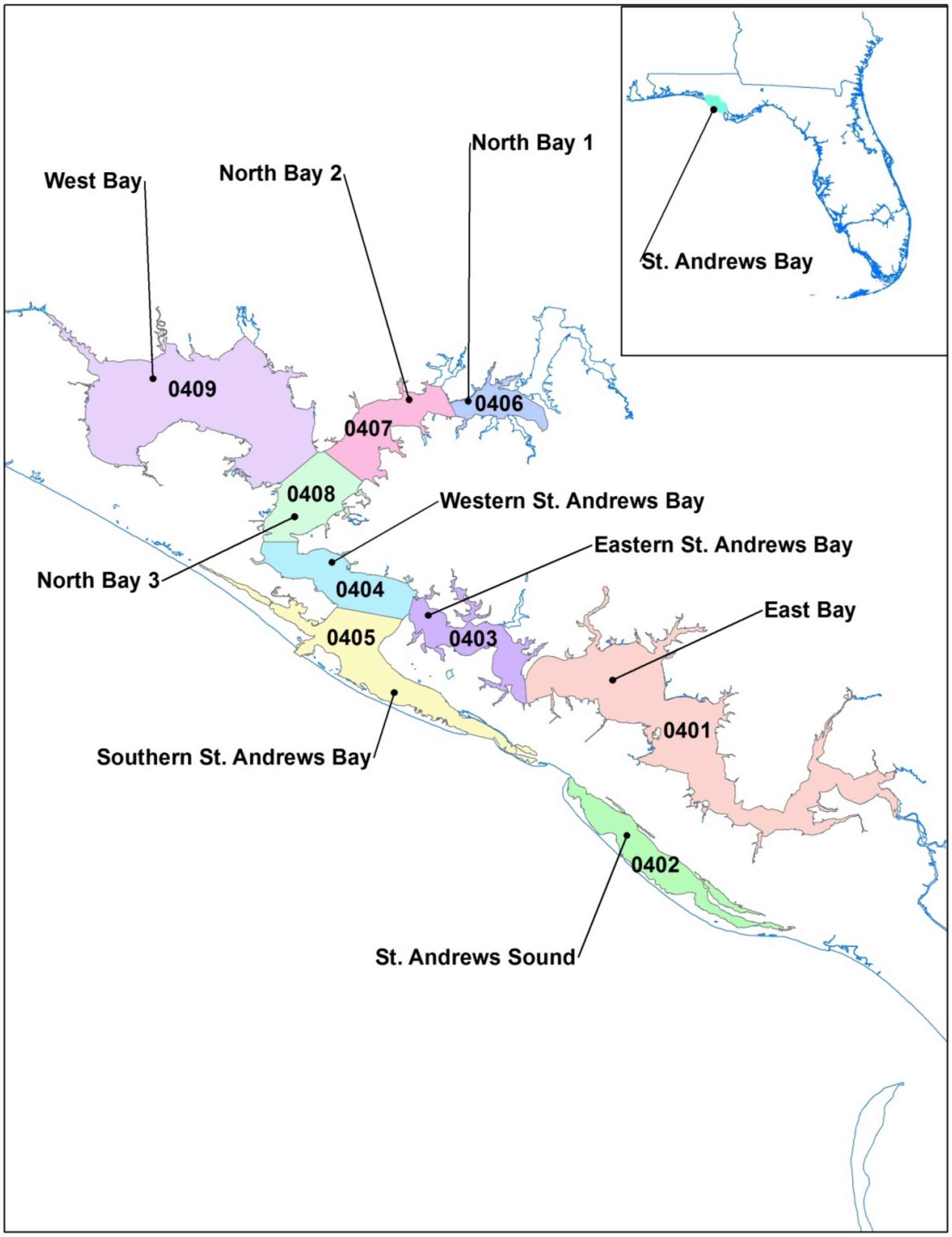

Figure 2-11. Results of St. Andrews Bay segmentation

2.4.5. Water Quality Targets

2.4.5.1. Seagrass Depth and Water Clarity Targets

Seagrass depth of colonization (measured as Z_c) and water clarity (measured as K_d) targets were established for estuary segments (Table 2-29) by analyzing the 1953 seagrass coverage for St. Andrews Bay (Figure 2-12) (USGS 2012). The depth of colonization targets were consistent with estimates of K_d(CDOM). Depth of colonization targets were fully supported. Depth of colonization targets could not be established for St. Andrews Sound (segment 0402) because bathymetric data could not be located for that part of the system.

Table 2-29. St. Andrews Bay seagrass depth and water clarity targets by segment

Estuary Segment	Depth of Colonization (Z_c) Target (m)	Light Attenuation Coefficient (K_d) Target (1/m)
0401	1.6	1.0
0402	No target	-
0403	2.3	0.7
0404	2.7	0.6
0405	3	0.5
0406	1.2	1.3
0407	1.6	1.0
0408	1.9	0.8
0409	1.5	1.1

Figure 2-12. Map of the 1953 seagrass distribution in St. Andrews Bay. The boundaries of EPA estuary segments are plotted in light grey.

2.4.5.2. Chlorophyll a *Target*

To prevent nuisance algal blooms and protect the estuary's designated uses, chl-a levels must not exceed 20 µg/L more than 10 percent of the time. The rationale for that target is provided in Section 1.2.2.

2.4.5.3. Dissolved Oxygen Targets

On the basis of the rationale that sufficient DO is necessary to protect aquatic life, as described in Section 1.2.3, the following DO targets were established:

- Minimum allowable DO of 4.0 mg/L as a water column average for each estuary segment 90 percent of the time over the simulation's time span
- Daily average DO of 5.0 mg/L as a water column average for each estuary segment 90 percent of the time over the simulation's time span
- Minimum 3-hour average DO of 1.5 mg/L in the bottom two layers for each estuary segment over the simulation's time span

2.4.6. Results of Analyses

2.4.6.1. Mechanistic Model Analysis

Average load contributions from the St. Andrews watershed are shown in Table 2-30.

Table 2-30. Average load contributions to St. Andrews Bay from the St. Andrews watershed (2002–2009)

	TN Load[1] (kg/d; Mean ± se)			TP Load[1] (kg/d; Mean ± se)		
Year	Existing[2]	Background[3]	Anthropogenic[4]	Existing[2]	Background[3]	Anthropogenic[4]
2002	1406 ± 59	1,063 ± 45	343 ± 31	54 ± 3	38 ± 2	15 ± 1
2003	3,690 ± 246	3,207 ± 225	482 ± 42	108 ± 5	81 ± 4	27 ± 1
2004	1,572 ± 112	1,299 ± 86	273 ± 37	53 ± 3	39 ± 2	14 ± 1
2005	2,145 ± 120	1,833 ± 106	312 ± 30	69 ± 4	51 ± 3	18 ± 1
2006	544 ± 38	334 ± 18	209 ± 28	25 ± 2	17 ± 2	8 ± 1
2007	951 ± 54	713 ± 44	238 ± 27	37 ± 3	26 ± 2	11 ± 1
2008	2,040 ± 194	1,693 ± 182	347 ± 33	67 ± 5	48 ± 4	19 ± 2
2009	1,582 ± 221	1,318 ± 208	264 ± 29	51 ± 5	37 ± 4	14 ± 1

[1] Annual average daily load from all terminal reaches
[2] Annual average daily load from all terminal reaches computed from existing model scenario
[3] Annual average daily load from all terminal reaches computed from background model scenario (remove anthropogenic sources/land use; use existing hydrology)
[4] Anthropogenic load computed as difference between existing and background load

The chl-a and water clarity targets were met for all St. Andrews Estuary segments according to 2002–2009 nutrient loads.

The DO targets were met using the 2002–2009 nutrient loads, except for the daily water column daily average target of 5 mg/L, which could not be met for segments 0401, 0402, 0406, and 0409 using either the 2002–2009 nutrient loads scenario or the non-anthropogenic nutrient scenario. Table 2-31 identifies which targets were met under 2002–2009 nutrient loads and which targets were not met.

An evaluation of model sensitivity to the water quality targets applied revealed that the DO target was insensitive to changes in nutrients in St. Andrews Bay.

Table 2-31. Water quality targets met for St. Andrews Estuary based on mechanistic modeling

Segment	DO	Chl-a	K_d
0401	No	Yes	Yes
0402	No	Yes	No target
0403	Yes	Yes	Yes
0404	Yes	Yes	Yes
0405	Yes	Yes	Yes
0406	No	Yes	Yes
0407	Yes	Yes	Yes
0408	Yes	Yes	Yes
0409	No	Yes	Yes

A summary of candidate criteria based on 2002–2009 nutrient loads for St. Andrews Estuary segments is given in Table 2-32.

Table 2-32. Summary of candidate criteria for St. Andrews Estuary derived from mechanistic modeling

Segment	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
0401	0.31	0.014	4.6
0402	0.14	0.009	2.3
0403	0.24	0.021	3.9
0404	0.19	0.016	3.1
0405	0.15	0.013	2.6
0406	0.22	0.012	3.7
0407	0.22	0.014	3.7
0408	0.21	0.016	3.4
0409	0.23	0.022	3.8

2.4.6.2. Statistical Model Analysis

Sufficient data were not available to conduct statistical model analysis in every segment in St. Andrews Bay. However, in segments where data were available, annual geometric mean light attenuation coefficient increased strongly with increased chl-a (Figure 2-13). In all segments except for segment 0409, the derived chl-a criteria protective of the water clarity endpoint was greater than the upper bound of observed chl-a values, and candidate chl-a criteria were based on the upper bound.

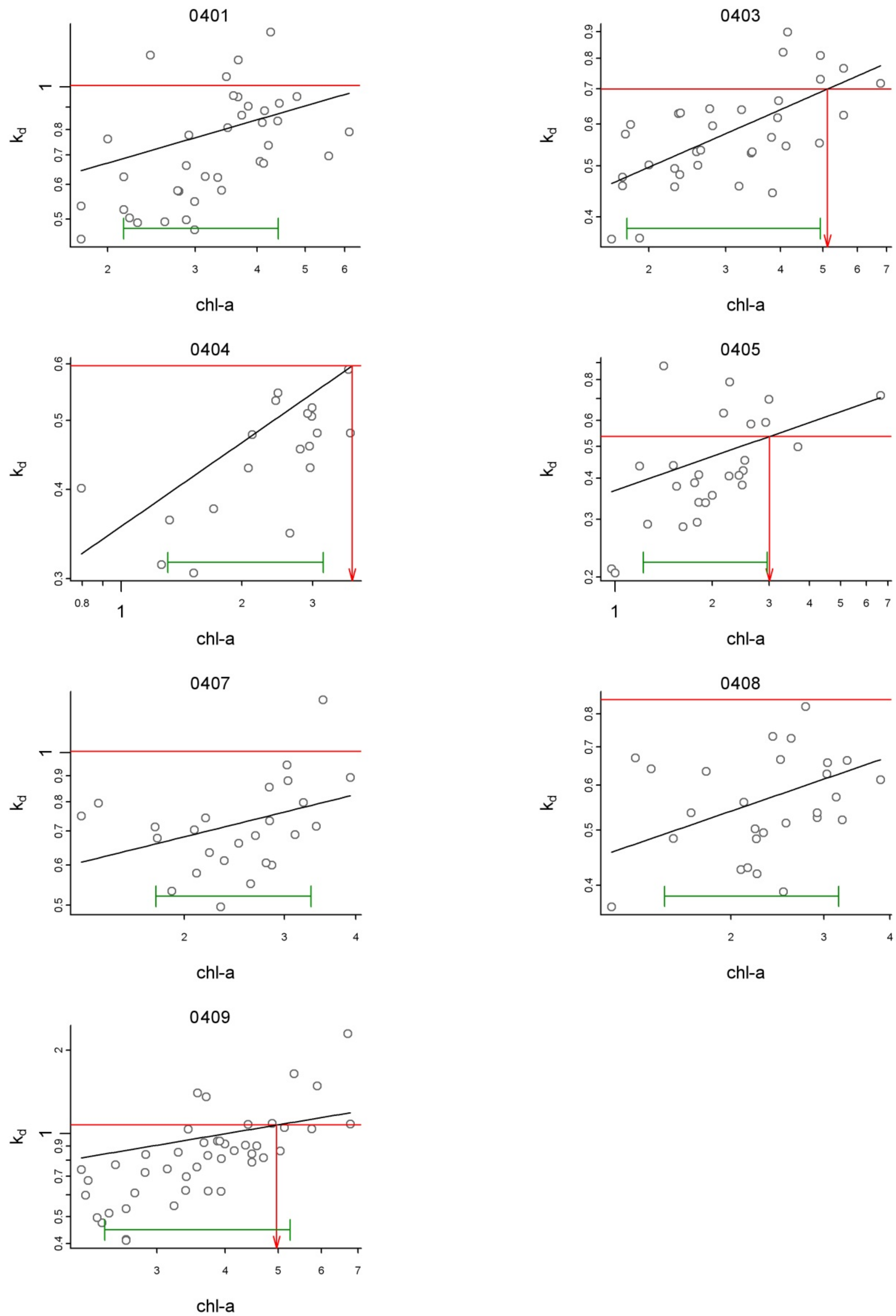

Figure 2-13. Relationships between annual geometric K_d and chl-a in St. Andrews Bay. Solid black line: segment-wide relationship; red horizontal line: K_d target; red vertical arrow: chl-a concentrations associated with K_d target; green line segment: 5th to 95th percentile range of chl-a concentrations, open circles: observed annual geometric mean K_d and chl-a concentrations.

Concentrations of chl-a did not exceed 20 µg/L more frequently than 10 percent in any of the segments of St. Andrews Bay. Hence, candidate chl-a criteria associated with the phytoplankton bloom endpoint were based on the upper bound of the data for all segments. Based on these analyses, candidate chl-a criteria were derived that protected both the water clarity and phytoplankton bloom endpoints (Table 2-33).

Table 2-33. Summary of candidate chl-a criteria in St. Andrews Bay. Sufficient data were not available in segments 0402 and 0406, so no chl-a criteria were calculated. Values with asterisks indicate that the predicted candidate criterion was greater than the upper bound of chl-a values, or less than the lower bound of chl-a values used in estimating the empirical relationship, so listed criterion is based on the upper or lower bound of the data.

Segment	Chl-a (clarity) (µg/L)	Chl-a (bloom) (µg/L)	Chl-a (final) (µg/L)
0401	4.4*	4.3*	4.3
0402	-	-	-
0403	4.9*	4.9*	4.9
0404	3.2*	3.0*	3.0
0405	3.0*	5.5*	3.0
0406	-	-	-
0407	3.3*	3.2*	3.2
0408	3.2*	3.3*	3.2
0409	5.0	5.8*	5.0

Relationships between annual geometric mean TN, TP, and chl-a were estimated and used to derive TN and TP criteria associated with candidate chl-a criteria (Figure 2-14 and Figure 2-15). Chl-a concentrations increased strongly with increasing TN and TP across all stations and all segments. For both TN and TP, derived candidate criteria values were greater than the upper bound of observed values in all segments except for segment 0405; candidate criteria are based on this upper bound. In segment 0405, TN and TP candidate criteria values corresponds with the point at which segment average predicted chl-a is the same as the chl-a target (Table 2-34).

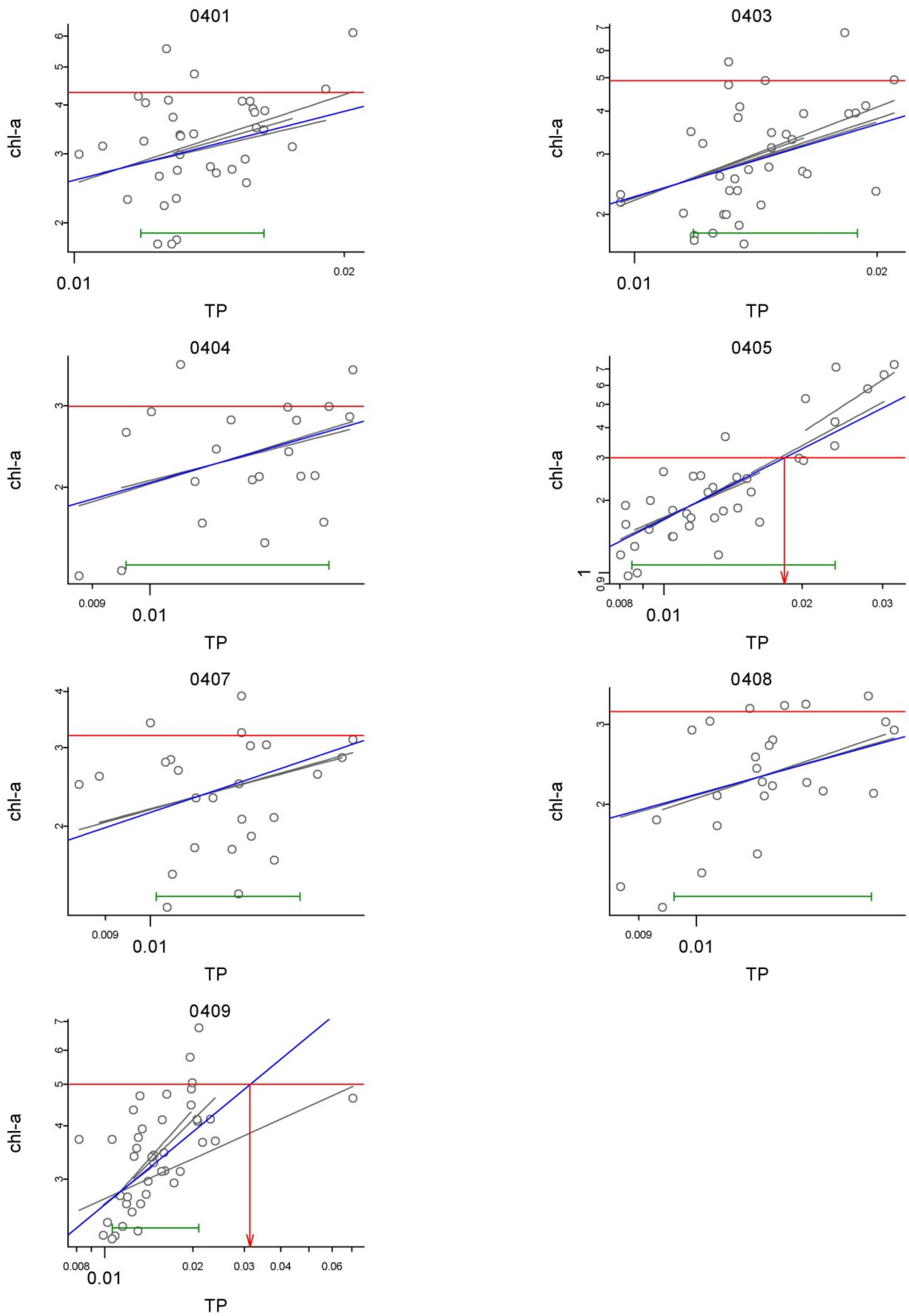

Figure 2-14. Relationships between TP and chl-a in St. Andrews Bay. Open circles: observed annual average values of TP and chl-a, red horizontal line: chl-a criterion, red vertical arrow: TP criterion associated with chl-a criterion, blue line: estimated segment-wide relationship between TP and chl-a, grey lines: estimated station-specific relationships between TP and chl-a, green line segment: 5th to 95th percentile range of observed annual geometric mean TP concentrations.

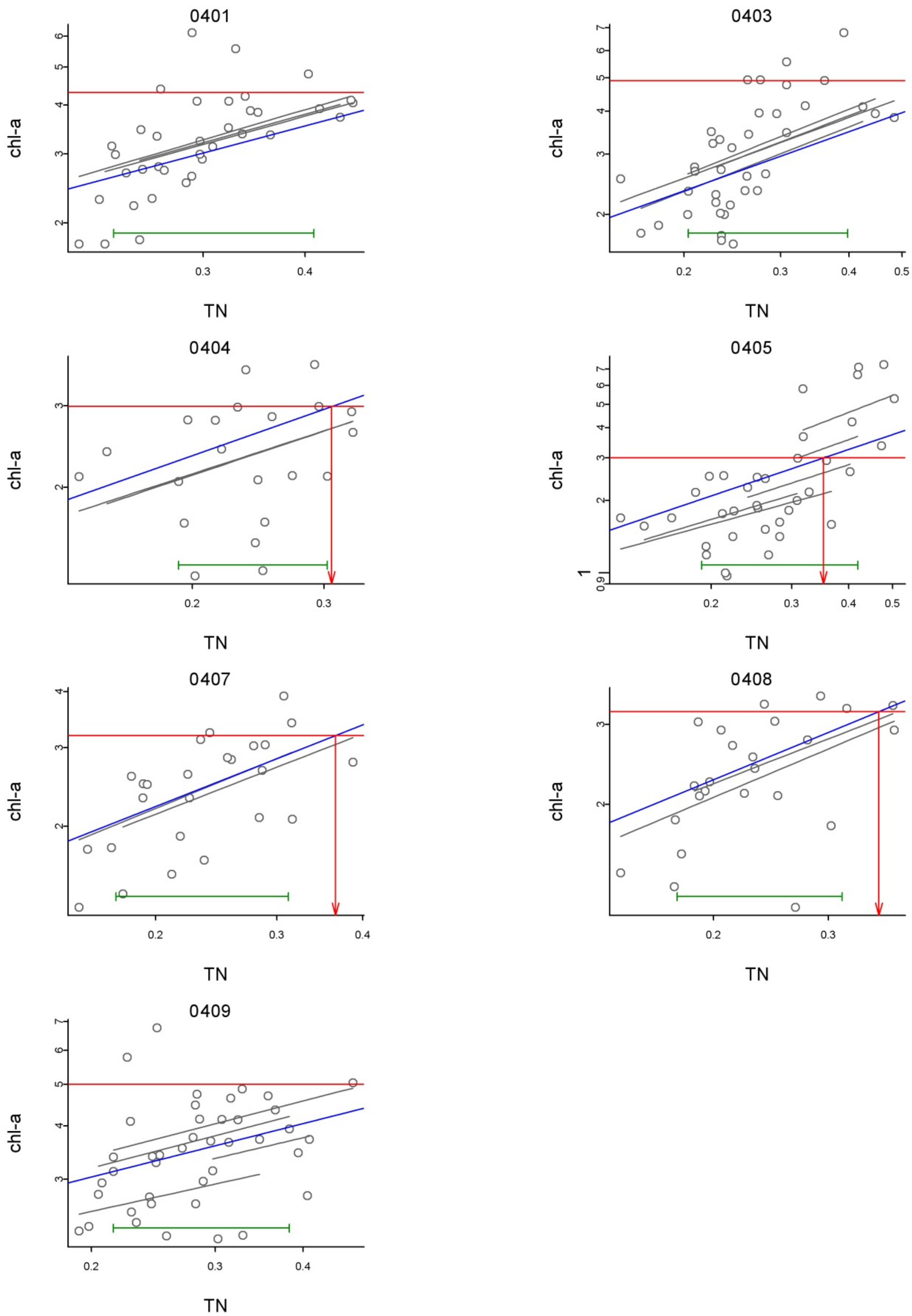

Figure 2-15. Relationships between TN and chl-a in St. Andrews Bay. Open circles: observed annual average values of TN and chl-a, red horizontal line: chl-a criterion, red vertical arrow: TN criterion associated with chl-a criterion, blue line: estimated segment-wide relationship between TN and chl-a, grey lines: estimated station-specific relationships between TN and chl-a, green line segment: 5th to 95th percentile range of observed annual geometric mean TN concentrations.

Table 2-34. Summary of candidate TN and TP criteria in St. Andrews Bay. No data were available in segments 0402 and 0406. Values with asterisks indicate that the predicted candidate criterion was greater than the upper bound of observed values, or less than the lower bound of observed values used in estimating the empirical relationship, so listed criterion is based on the upper or lower bound of the data.

Segment	TN (mg/L)	TP (mg/L)
0401	0.41*	0.016
0403	0.40*	0.019
0404	0.30*	0.014
0405	0.35	0.018
0407	0.31*	0.014
0408	0.32*	0.014
0409	0.38*	0.021

2.4.7. Application of Analyses for Proposed Numeric Nutrient Criteria

EPA evaluated various lines of evidence for deriving numeric nutrient criteria in St. Andrews Bay. Data necessary to conduct statistical analyses were not available for all segments of St. Andrews Bay. However, the mechanistic model provided values for every segment in the estuary. As a result, EPA derived the proposed numeric nutrient criteria for St. Andrews Bay shown in the table below based on the mechanistic modeling results.

EPA evaluated three endpoints in the mechanistic modeling approach for St. Andrews Bay: (1) water clarity protective of historic depth of seagrasses, (2) chl-a concentrations associated with balanced phytoplankton biomass, and (3) DO concentrations sufficient to maintain aquatic life. EPA found that the DO target was not met under calibrated 2002–2009 nutrient loads and was insensitive to changes in nutrients. Therefore, the DO endpoint was not used in St. Andrews Bay. However, both the water clarity and chl-a targets were met under the calibrated 2002–2009 nutrient loads and were sensitive to changes in nutrients. The proposed criteria were derived to be protective of water clarity and chl-a concentrations. The values under mechanistic modeling represent the 90th percentile annual geometric mean nutrient concentrations from the 2002–2009 modeled nutrient loads.

Statistical modeling analyses provided numeric nutrient criteria for several segments in St. Andrews Bay, and of these segments, analyses of dilution relationships indicated that segment 0407 was the controlling segment for the statistically derived criteria. Hence, comparisons between criteria derived using mechanistic and statistical models are best conducted within this segment. In segment 0407, statistically derived criteria values are very similar to those derived using mechanistic models, and corroborate the final proposed criteria.

Since the mechanistic model provided values for every segment in the estuary, EPA derived the proposed numeric nutrient criteria for St. Andrews Bay shown in the table below based on the mechanistic modeling results.

Proposed numeric nutrient criteria for TN, TP, and chl-a in St. Andrews Bay segments are summarized in Table 2-35.

Table 2-35. Proposed and candidate numeric nutrient criteria for St. Andrews Bay segments

		Proposed Criteria			Mechanistic Modeling			Statistical Modeling		
SEGMENT	SEGMENT ID	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
East Bay	0401	0.31	0.014	4.6	0.31	0.014	4.6	0.41	0.016	4.3
St. Andrews Sound	0402	0.14	0.009	2.3	0.14	0.009	2.3	–	–	–
Eastern St. Andrews Bay	0403	0.24	0.021	3.9	0.24	0.021	3.9	0.40	0.019	4.9
Western St. Andrews Bay	0404	0.19	0.016	3.1	0.19	0.016	3.1	0.30	0.014	3.0
Southern St. Andrews Bay	0405	0.15	0.013	2.6	0.15	0.013	2.6	0.35	0.018	3.0
North Bay 1	0406	0.22	0.012	3.7	0.22	0.012	3.7	–	–	–
North Bay 2	0407	0.22	0.014	3.7	0.22	0.014	3.7	0.31	0.014	3.2
North Bay 3	0408	0.21	0.016	3.4	0.21	0.016	3.4	0.32	0.014	3.2
West Bay	0409	0.23	0.022	3.8	0.23	0.022	3.8	0.38	0.021	4.4

2.4.8. Downstream Protective Values

In St. Andrews Bay, mechanistic models were applied to derive proposed DPVs for TN and TP shown in Table 2-36.

Table 2-36. Proposed DPVs for St. Andrews Bay

Tributary[a]	LSPC model watershed/ Structure ID	USGS Reach Code	Estuary Segment	TN (mg/L)	TP (mg/L)
Wetappo Creek	40001	03140101001347	0401	0.50	0.016
	40002	03140101001035	0401	0.60	0.023
	40005	03140101006255	0402	0.61	0.020
Beatty Bayou	40007	03140101001000	0406	0.82	0.032
Cooks Bayou	40008	03140101001034	0401	0.58	0.017
Boggy Creek	40011	03140101001032	0401	0.61	0.017
Horseshoe Creek	40017	03140101000701	0401	0.56	0.012
Pitts Bayou	40018	03140101003238	0403	0.85	0.037
Sandy Creek	40022	03140101001332	0401	0.53	0.015
	40023	03140101001001	0408	0.94	0.030
	40025	03140101002467	0408	0.88	0.027
	40036	03140101006225	0407	0.63	0.032
	40041	03140101001029	0403	0.80	0.032
	40042	03140101001028	0403	0.90	0.028
Ward Creek	40053	03140101003488	0409	0.76	0.022
	40054	03140101000691	0409	0.53	0.017
Crooked Creek	40056	03140101000696	0409	0.54	0.014
Burnt Mill Creek	40057	03140101003512	0409	0.52	0.017
Alligator Bayou	40059	03140101006233	0407	0.64	0.016
Mill Bayou	40066	03140101006243	0407	0.73	0.030
	40067	03140101000495	0404	0.93	0.029
Farmdale Bayou	40068	03140101006383	0401	0.63	0.020
	40070	03140101002622	0405	0.89	0.030
	40071	03140101001024	0407	0.87	0.029
	40076	03140101003239	0401	0.68	0.026
Deer Point Lake	Deer Point Lake Dam	03140101006341	0406	0.14	0.008

[a] Tributary names left blank are unnamed

2.4.9. References

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2.5. St. Joseph Bay

2.5.1. Proposed Numeric Nutrient Criteria

Proposed numeric nutrient criteria for TN, TP, and chl-a in St. Joseph Bay are summarized in Table 2-37.

Table 2-37. Proposed numeric nutrient criteria for St. Joseph Bay

Segment Name	Segment Number	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
St. Joseph Bay	0501	0.25	0.018	3.8

2.5.2. General Characteristics

2.5.2.1. System Description

St. Joseph Bay is a coastal lagoon in Gulf County, semi-enclosed by the St. Joseph Peninsula and adjacent to the St. Andrews Bay and Apalachicola watersheds (FDEP 2010).[30] St. Joseph Bay stretches approximately 15 mi (24 km) from north to south and spans 6 mi (10 km) at its widest point (FDEP 2008, 2010). In the eastern Gulf of Mexico, St. Joseph Bay is one of the only embayments without a major source of surface freshwater inflow (Heck et al. 2000; NWFWMD 2000). Approximately 79 percent of the land use in the St. Joseph Bay Basin is water and wetlands (FDEP 2010). St. Joseph Bay is a high-salinity coastal lagoon with an average salinity of 35 PSU (FDEP 2008). The bay is considered part of the St. Andrews Bay watershed, the only major estuarine drainage basin entirely within the Florida Panhandle (NWFWMD 2000). The total surface area of the bay at mean high water is approximately 69 mi^2 (178 km^2 [43,872]) (FDEP 2011a). The overall mean depth of the bay is 21 ft (6.4 m); the shallowest parts of the bay have an average depth of 3 ft (0.9 m), and the deepest parts are approximately 35 ft (11 m); the majority of the bay has a shallow shoreline (FDEP 2008).

In 1969 St. Joseph Bay was designated as a 114 mi^2 (295 km^2 [73,000 ac]) aquatic preserve. The St. Joseph Bay State Buffer Preserve was created in 1995 to protect and conserve a regionally significant natural area of approximately 7.8 mi^2 (20 km^2 [5,018 ac]) with outstanding ecological, economic, historical, and cultural values (FDEP 2008).

Gulf County has warm, humid summers and mild winters. Wind conditions are typically northerly during winter and southerly during summer. Peak rainfall periods occur primarily in the summer and fall with total average annual rainfall around 60 in (152 cm). The average high temperature is 79 °F (26 °C), and the low temperature is approximately 55 °F (13 °C) (FDEP 2008). Water temperatures in the bay range from 45 to 91 °F (7 to 33 °C), with the lowest temperatures occurring in December and January, and the highest temperatures occurring in July and August (FDEP 2010).

St. Joseph Bay is on an offshore extension of the Gulf Coast Lowlands geomorphic province (FDEP 2008). The land in Gulf County, surrounding the bay, is divided into three distinct, open coast segments (Foster and Cheng 2001). The sediment underlying St. Joseph Bay is primarily composed of clean quartz sand and biological carbonates. Detrital sands travel from the eastern Apalachicola River in longshore drifts and are deposited in this area. Average organic content of the sediment is approximately 1.4 percent, with an average organic carbon/organic nitrogen ratio of 15.4 (Stewart and Gorsline 1962).

Gulf County has a total area of 745 mi^2 (1,930 km^2) (US Census Bureau 2011a). The major urban area on St. Joseph Bay is the City of Port St. Joe with a population of 3,445 (FDEP 2006; US Census Bureau 2010). In the mid-1990s, St. Joseph Peninsula State Park saw a 50 percent increase in the number of annual visitors (FDEP 2008). Gulf County has experienced a 19 percent increase in population from 2000 to 2010 (US Census Bureau 2011b).

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[30] The information presented in this system description was compiled to summarize local information pertaining to the proposed numeric nutrient criteria for this estuary. For more information on EPA's process of delineating, segmenting, and deriving numeric nutrient criteria for this estuary, please see Section 1.3.

2.5.2.2. Impaired Waters[31]

No Class II or Class III marine WBIDs with a nutrient-related parameter are on Florida's CWA section 303(d) list approved by EPA in the St. Joseph Bay area.[32] No Class II or Class III marine WBIDs with nutrient-related TMDLs were documented for this region.[33]

2.5.2.3. Water Quality

As part of the Gulf of Mexico Program, EPA has designated St. Joseph Bay as a Gulf of Mexico Ecological Management Site (FDEP 2008). In addition, FDEP designated St. Joseph Bay Aquatic Preserve as an OFW.[34]

Most water quality data for St. Joseph Bay were collected by the Water Quality Monitoring Program at St. Joseph Bay Aquatic Preserve, a collaboration of the preserve and University of Florida's LakeWatch program (FDEP 2011b). The various sources of nutrients to St. Joseph Bay include nonpoint sources (septic tanks and stormwater), point sources (wastewater treatment plants [WWTP] and industrial), atmospheric deposition, and natural sources (plant leaf litter) (FDEP 2010).

DO saturations in St. Joseph Bay average 98 percent, and hypoxia has not been observed (FDEP 2010). During a study in 1997 and 1998, mean DO concentrations were comparable in the St. Joseph Bay and the Gulf County Canal at 8.3 and 7.6 mg/L, respectively (Berndt and Franklin 1999).

According to data collected by FDEP's Office of Coastal and Aquatic Managed Areas, St. Joseph Bay Aquatic Preserve, and LakeWatch/Project COAST between 2001 and 2009 (from eight stations throughout the bay), chl-a concentrations in the bay were low, with annual geometric means ranging from 1.1 to 3.2 µg/L, and a long-term geometric mean of 2.0 µg/L. Under non-storm event conditions, the mean turbidity was 0.94 NTU (FDEP 2010).

From 2001 to 2009, TN concentrations in the bay were relatively low and stable; annual geometric means ranged from 0.192 to 0.278 mg/L (FDEP 2010). TP concentrations were relatively low and stable from 2001 to 2009, with fluctuations of only a few micrograms per liter (FDEP 2010).

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[31] For more information about the data source, see Volume 1, Appendix A.

[32] The nutrient-related 303(d) list was a compilation of EPA-approved 303(d) listing information for nutrients, chl-a, and DO provided in three decision documents: September 2, 2009, Basin Groups 1, 2, and 5 EPA Decision Document (http://www.epa.gov/region4/water/tmdl/florida/documents/fl09303d_decisiondoc_090209.pdf);
May 13, 2010, Basin Group 3 EPA Decision Document (http://www.epa.gov/region4/water/tmdl/florida/documents/fl303d_%20partialapproval_decision_docs051410.pdf);
and the December 21, 2010, Basin Group 4 EPA Decision Document (http://www.epa.gov/region4/water/tmdl/florida/documents/group_4_final_dec_doc_and_partial_app_letter_12_21_10.pdf).

[33] TMDLs were identified in February 2011 by compiling nutrient-related draft/final TMDLs from the following three sources:
FDEP TMDL website (http://www.dep.state.fl.us/water/tmdl/index.htm)
EPA Region 4 website (http://www.epa.gov/region4/water/tmdl/florida/index.html)
EPA National WATERS expert query tool (http://www.epa.gov/waters/tmdl/expert_query.html).

[34] Section 62-302.700, F.A.C.

2.5.2.4. Biological Characteristics

The Florida Panhandle is home to many rare species. St. Joseph Bay supports a bountiful and biologically diverse ecosystem of seagrasses, salt marshes, benthic communities, dolphins, sharks, sea turtles, commercial and recreational fish, and various bird species. Within the panhandle, St. Joseph Peninsula is an important habitat for nesting loggerhead sea turtles (*Caretta caretta*) (FDEP 2008).

Approximately 30 percent of the area in St. Joseph Bay basin (39 mi^2 [102 km^2]) consists of wetlands (FDEP 2010). Wetlands are found chiefly along the shores of the bay and help control flooding and erosion, remove and retain excessive nutrients (such as nitrogen and phosphorus), and provide vital habitat for terrestrial and aquatic wildlife including shrimp, fish, crabs, waterfowl, wading birds, and mammals. The entire shoreline of St. Joseph Bay is bordered by salt marsh habitat, covering approximately 1.2 mi^2 (3.1 km^2 [762.58 ac]) in the area of the St. Joseph Bay Aquatic Preserve. Dominant species of seagrass found in St. Joseph Bay salt marshes are black needlerush (*Juncus roemerianus*) and smooth cordgrass (*Spartina alterniflora*) (FDEP 2008).

St. Joseph Bay contains one of the most abundant and richest concentrations of marine grasses along Florida's northwest coast. Seagrass is considered the key ecological species in the St. Joseph Bay area (FDEP 2010). Five species of seagrasses have been observed in the seagrass meadows that are present at the bottom of St. Joseph Bay, including Cuban shoal grass (*Halodule wrightii*), manatee grass (*Syringodium filiforme*), turtle grass (*Thalassia testudinum*), widgeon grass (*Ruppia maritime*), and star grass (*Halophila engelmanni*) (FDEP 2008).

In the early 1900s, the bay's salt marshes, which serve as the exclusive habitat for various birds, mammals, invertebrates, and reptiles, became stressed and began to decline because of a recurring, unknown pathogen (FDEP 2008).

Dominant macroalgal species include star alga, *Argardhiella, Avrainvellea, Batophora, Bryopsis, Calothrix, Caulerpa, Chondria, Cladophora, Dictyota, Digenia, Gracilaria, Halimeda, Laurencia, Oscillatoria*, shaving brush, *Rhipocephalus*, and *Sargassum*. Algal beds are categorized with many marine or estuarine natural communities including seagrass beds, tidal marsh, and tidal swamps. Dredge and fill activities that physically remove or bury the beds are the primary threat to marine and estuarine algal beds. Algal beds provide critical habitat for the commercially important juvenile spiny lobster (FDEP 2008).

The estimated productivity of macroinvertebrates in St. Joseph Bay seagrass habitats is the highest among those reported in other literature (Valentine and Heck 1993). Much of the productivity in the area is attributed to the nearshore salt marsh and seagrass habitats (Duke and Kruczynski 1992). In St. Joseph Bay, stone crabs (*Menippe mercinaria*), bay scallops (*Argopecten irradians*), horse conchs (*Pleuroploca gigantean*), lightening whelks (*Busycon perversum pulleyi*), and pen shells (*Atrina rigida*) occur in greater numbers than other places in the northern Gulf of Mexico (FDEP 2008). Pen shells and bay scallops are abundant in St. Joseph Bay. Rich communities of both motile and sessile species of pen shells provide a source of hard substrate for a number of organisms to attach (FDEP 2008; Munguia 2004). This area also hosts one of the healthiest bay scallop populations in Florida, according to recruitment rates (FDEP 2008).

St. Joseph Bay is home to many species of value to the commercial and recreational fishing industry. Gray snapper (*Lutjanus griseus*), black sea bass (*Centropristis striata*), and yellowtail snapper (*Ocyrus chrusurus*) are a few species that contribute to the value of the commercial fishing industry. Anglers have access to a variety of species, including spotted seatrout (*Cynoscion nebulosus*), king mackerel (*Scomberomorus cavalla*), Spanish mackerel (*S. maculatus*), red drum (*Sciaenops ocellatus*), southern flounder (*Paralichthys lethostigma*), tarpon (*Megalops atlanticus*), and mullet (*Mugil cephalus, M. curema*). In Gulf County, the recreational fishing industry is a valuable source of revenue (FDEP 2008). Shellfish harvesting is also an important activity (FDEP 2008), and elevated concentrations of red tide organisms (e.g., *Karenia brevis*) have caused periodic closing of shellfish harvesting in St. Joseph Bay (FDEP 2010).

For a more detailed summary of this water body, see Volume 1, Appendix A.

2.5.3. Data Used

One data source specific to St. Joseph Bay was used in addition to those sources common to all estuary systems (described in Section 1.4.3), as summarized in Table 2-38.

Table 2-38. Data source specific to St. Joseph Bay models

Data	Source	Location Used
Springs discharge and water quality data	Florida Department of Environmental Protection (FDEP 2004, 2009)	St. Andrews watershed model

2.5.4. Segmentation

St. Joseph Bay is a lagoonal embayment connected to St. Andrews Bay by the Gulf Intracoastal Waterway. The salinity distribution within the lagoon is relatively homogeneous with a range of 30–32 PSU on average. Hence, EPA has determined that further segmentation is not necessary, as shown in Figure 2-16.

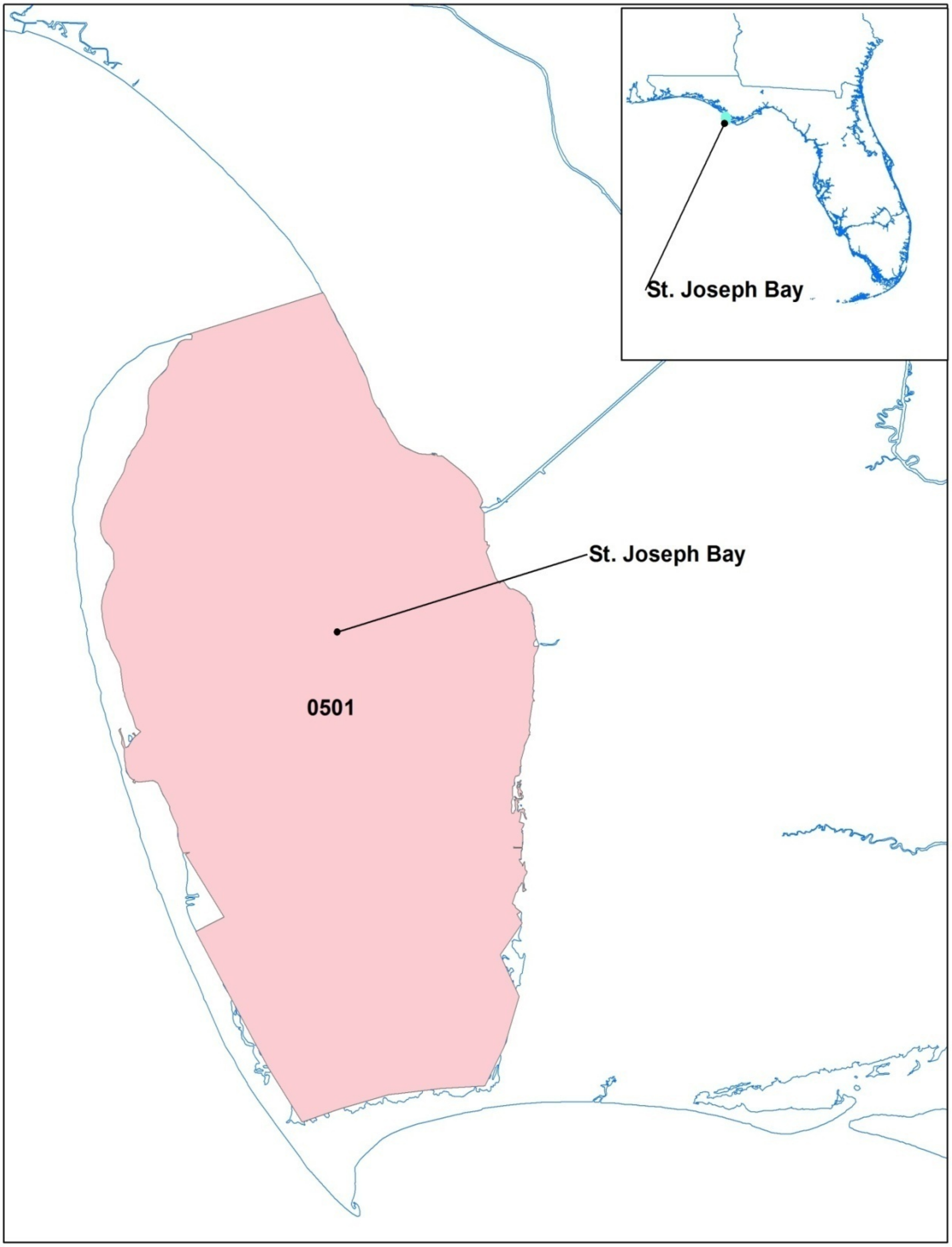

Figure 2-16. Results of St. Joseph Bay segmentation

2.5.5. Water Quality Targets

2.5.5.1. Seagrass Depth and Water Clarity Targets

A seagrass depth of colonization (measured as Z_c) target and a water clarity (measured as K_d) target (Table 2-39) were established for the St. Joseph Bay Estuary segment (0501) based on the 1992 and 2010 seagrass coverages (Figure 2-17) (FWRI 2012; USGS 2012). Seagrass depth of colonization was highest in 1992.

Table 2-39. St. Joseph Bay seagrass depth of colonization and water clarity targets

Estuary segment	Depth of Colonization (Z_c) Target (m)	Light Attenuation Coefficient (K_d) Target (1/m)
0501	1.6	1.0

Figure 2-17. 1992 seagrass coverage in St. Joseph Bay

2.5.5.2. Chlorophyll a *Target*

To prevent nuisance algal blooms and protect the estuary's designated uses, chl-a levels must not exceed 20 µg/L more than 10 percent of the time. The rationale for that target is provided in Section 1.2.2.

2.5.5.3. Dissolved Oxygen Targets

On the basis of the rationale that sufficient DO is necessary to protect aquatic life, as described in Section 1.2.3, the following DO targets were established:

- Minimum allowable DO of 4.0 mg/L as a water column average for each estuary segment 90 percent of the time over the simulation's time span
- Daily average DO of 5.0 mg/L as a water column average for each estuary segment 90 percent of the time over the simulation's time span
- Minimum 3-hour average DO of 1.5 mg/L in the bottom two layers for each estuary segment over the simulation's time span

2.5.6. Results of Analyses

2.5.6.1. Mechanistic Model Analysis

Average load contributions to St. Joseph Bay from the St. Andrews watershed are shown in Table 2-40.

Table 2-40. Average load contributions to St. Joseph Bay from the St. Andrews watershed (2002–2009)

	TN Load[1] (kg/d; Mean ± se)			TP Load[1] (kg/d; Mean ± se)		
Year	Existing[2]	Background[3]	Anthropogenic[4]	Existing[2]	Background[3]	Anthropogenic[4]
2002	133 ± 6	99 ± 4	34 ± 3	5 ± 0	4 ± 0	1 ± 0
2003	331 ± 28	287 ± 26	45 ± 4	9 ± 1	7 ± 0	2 ± 0
2004	146 ± 14	119 ± 11	27 ± 4	5 ± 0	4 ± 0	1 ± 0
2005	193 ± 12	163 ± 11	30 ± 3	6 ± 0	4 ± 0	1 ± 0
2006	56 ± 4	35 ± 2	21 ± 3	3 ± 0	2 ± 0	1 ± 0
2007	91 ± 5	67 ± 4	24 ± 3	4 ± 0	3 ± 0	1 ± 0
2008	188 ± 21	156 ± 20	33 ± 3	6 ± 1	4 ± 0	1 ± 0
2009	153 ± 28	127 ± 26	26 ± 3	5 ± 0	3 ± 0	1 ± 0

[1] Annual average daily load from all terminal reaches
[2] Annual average daily load from all terminal reaches computed from existing model scenario
[3] Annual average daily load from all terminal reaches computed from background model scenario (remove anthropogenic sources/land use; use existing hydrology)
[4] Anthropogenic load computed as difference between existing and background load

The chl-a and light attenuation coefficient (K_d) targets were met for the St. Joseph Bay segment according to 2002–2009 nutrient loads.

The DO targets were met according to 2002–2009 nutrient loads, except for the water column daily average target of 5 mg/L, which could not be met for segment 0501 using either the 2002–2009 nutrient loads scenario or the non-anthropogenic nutrient scenario. Table 2-41 identifies which targets were met under 2002–2009 nutrient loads and which targets were not met.

An evaluation of model sensitivity to the water quality targets applied revealed that the DO target was insensitive to changes in nutrients in St. Joseph Bay.

Table 2-41. Water quality targets met for St. Joseph Bay based on mechanistic modeling

Segment	DO	Chl-a	K_d
0501	No	Yes	Yes

A summary of candidate criteria based on the 2002–2009 nutrient loads for the St. Joseph Bay segment is given in Table 2-42.

Table 2-42. Summary of candidate criteria for St. Joseph Bay derived from mechanistic modeling

Segment	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
0501	0.20	0.010	2.9

2.5.6.2. Statistical Model Analysis

Analysis of empirical data collected from St. Joseph Bay indicated that chl-a concentrations associated with protecting both the phytoplankton bloom frequency and water clarity endpoints were greater than the upper bound range of observations (Figure 2-18 and Figure 2-19). Only four geometric mean chl-a values were available that were matched with geometric mean light attenuation coefficient values, a sample size that is not sufficient to estimate the upper bound of chl-a concentrations with enough precision. Hence, the candidate chl-a criterion is based on the upper bound of observed values used in estimating the bloom frequency relationship.

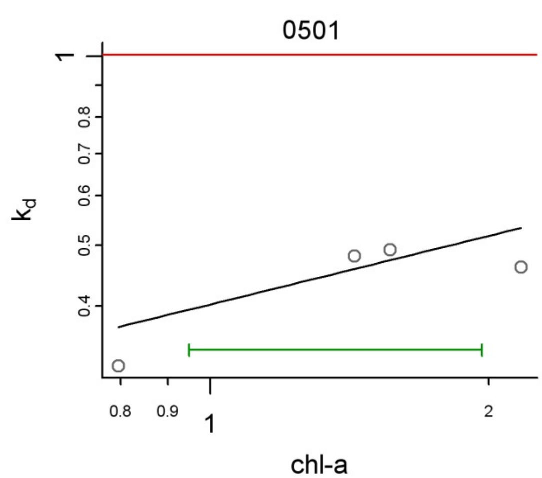

Figure 2-18. Modeled relationship between K_d and chl-a. Open circles: annual geometric means of K_d and chl-a, solid line: estimated mean relationship, red horizontal line: K_d target, vertical red arrow: criterion value associated with stressor-response relationship.

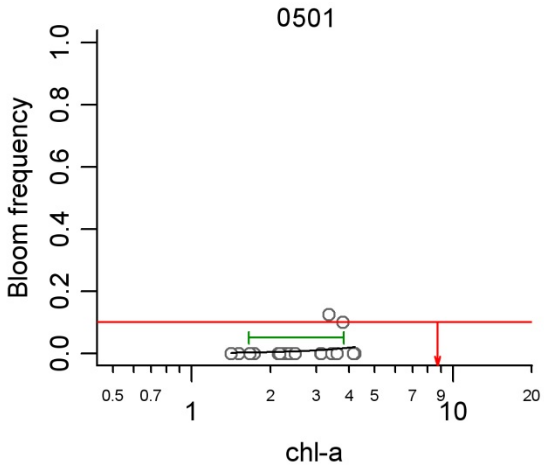

Figure 2-19. Modeled relationship between bloom frequency and chl-a. Solid black line: modeled mean relationship. Open circles: observed annual geometric means. Dashed horizontal line: 10 percent bloom frequency endpoint. Green line segment: 5th to 95th percentile range of observed data.

TN and TP concentrations that were associated with the candidate chl-a criterion were both greater than the upper bound of the range of observed TN and TP values, and therefore, candidate criteria are based on the upper bounds of the observed values (Figure 2-20).

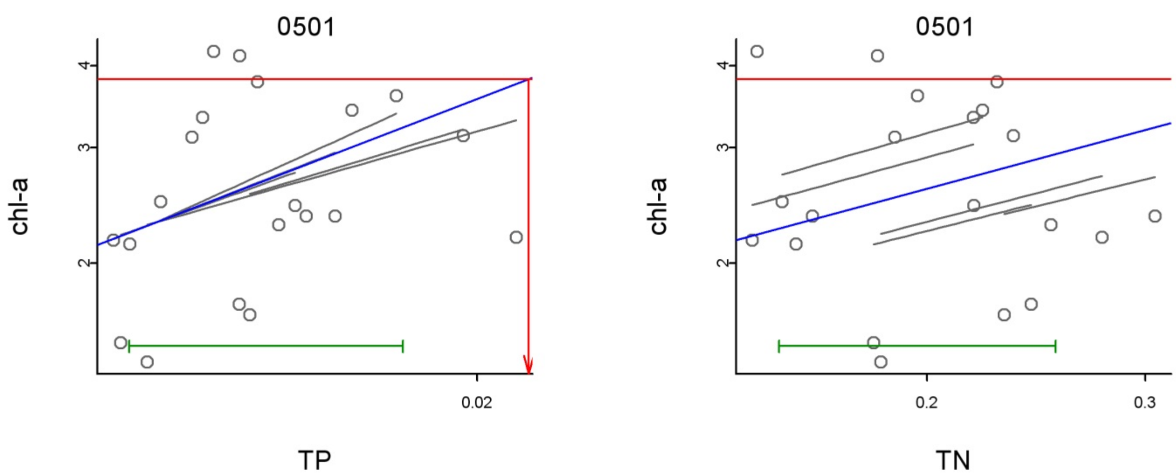

Figure 2-20. Relationships between TN, TP, and chl-a in St. Joseph Bay. Open circles: observed annual average values of TN and chl-a, red horizontal line: chl-a criterion, red vertical arrow: TN and TP criterion associated with chl-a criterion, blue line: estimated segment-wide relationship between TN, TP, and chl-a, grey lines: estimated station-specific relationships between TN, TP, and chl-a, green line segment: 5th to 95th percentile range of observed annual geometric mean TN and TP concentrations.

2.5.7. Application of Analyses for Proposed Numeric Nutrient Criteria

EPA evaluated various lines of evidence for deriving numeric nutrient criteria in St. Joseph Bay. Data were sufficient to use statistical analyses as the primary line of evidence when deriving criteria in St. Joseph Bay. EPA's analysis indicated that criteria values to meet the bloom prevention and water clarity targets were greater than the range of TN, TP, and chl-a concentrations observed in the data. Thus, proposed criteria were computed as estimates of the upper bound of observed annual average values of TN, TP, and chl-a.

EPA evaluated three endpoints in the mechanistic modeling approach for St. Joseph Bay: (1) water clarity protective of historic depth of seagrasses, (2) chl-a concentrations associated with balanced phytoplankton biomass, and (3) DO concentrations sufficient to maintain aquatic life. EPA found that the DO target was not met under the calibrated 2002–2009 nutrient loads and was insensitive to changes in nutrients. Therefore, the DO endpoint was not used in St. Joseph Bay. However, both the water clarity and chl-a targets were met under the calibrated 2002–2009 nutrient loads and were sensitive to changes in nutrients. The candidate criteria were derived to be protective of water clarity and chl-a concentrations. The values under mechanistic modeling represent the 90th percentile annual geometric mean nutrient concentrations from the 2002–2009 modeled nutrient loads.

In comparing the mechanistic and statistical model results, EPA found the results to be comparable and corroborative.

Proposed numeric nutrient criteria for TN, TP, and chl-a in the St. Joseph Bay segment are summarized in Table 2-43.

Table 2-43. Proposed and candidate numeric nutrient criteria for the St. Joseph Bay segment

		Proposed Criteria			Mechanistic Modeling			Statistical Modeling		
SEGMENT	SEGMENT ID	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
St. Joseph Bay	0501	0.25	0.018	3.8	0.20	0.010	2.9	0.25	0.018	3.8

2.5.8. Downstream Protective Values

In St. Joseph Bay mechanistic models were applied to derive proposed DPVs for TN and TP shown in Table 2-44.

Table 2-44. Proposed DPVs for St. Joseph Bay

Tributary[a]	LSPC Model Watershed ID	USGS Reach Code	Estuary Segment	TN (mg/L)	TP (mg/L)
	40045	03140101002395	0501	0.67	0.020
	40049	03140101000572	0501	0.68	0.024

[a] Tributary names left blank are unnamed

2.5.9. References

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2.6. Apalachicola Bay

2.6.1. Proposed Numeric Nutrient Criteria

Proposed numeric nutrient criteria for TN, TP, and chl-a in Apalachicola Bay segments are summarized in Table 2-45.

Table 2-45. Proposed numeric nutrient criteria for Apalachicola Bay segments

Segment Name	Segment Number	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
St. George Sound	0601	0.53	0.019	3.6
Apalachicola Bay	0602	0.51	0.019	2.7
East Bay	0603	0.76	0.034	1.7
St. Vincent Sound	0605	0.52	0.016	11.9
Apalachicola/Offshore	0606	0.30	0.008	2.3

2.6.2. General Characteristics

2.6.2.1. System Description

Apalachicola Bay is in the central part of the Florida Panhandle and has a surface area of 81 mi^2 (210 km^2). The shallow, east-west oriented estuarine system is commonly divided into four areas—St. Vincent Sound, East Bay, St. George Sound, and Apalachicola Bay.[35] However, delineation of the watershed often also includes Alligator Harbor and Dog Island Sound, which are discussed in a later section of this technical support document (TSD). The Apalachicola Bay system in Florida includes upland, floodplain, riverine, estuarine, and barrier island environments (ANERR 2008). The Apalachicola River Basin, a subwatershed that drains to Apalachicola Bay, is part of the larger Apalachicola-Chattahoochee-Flint River Basin (watershed), which drains about 20,000 mi^2 (51,800 km^2) in western Georgia, eastern Alabama, and the Florida Panhandle (FDEP 2005; Frick et al. 1998).

Apalachicola Bay is a highly productive estuary that provides habitat for estuarine species and supports the largest oyster harvesting industry in Florida and extensive shrimping, crabbing, and commercial fishing (FDEP 2005; Livingston 2008; NWFWMD 2007). Apalachicola Bay is part of the Apalachicola National Estuarine Research Reserve, which also extends to the lower half of the Apalachicola River, its floodplain, and areas of adjoining uplands (ANERR 2008; FDEP and ANERR 1998). Waters in the Apalachicola Bay basin designated as OFWs include Chipola River, Apalachicola River (portions), Apalachicola Bay, Apalachicola Bay Aquatic Preserve, Apalachicola National Estuarine Research Reserve, and Apalachicola National Forest.[36]

Although some fluctuation in seasonal and annual temperature occurs, particularly by elevation and proximity to the coast, temperatures in the Apalachicola Basin are generally mild, with a

[35] The information presented in this system description was compiled to summarize local information pertaining to the proposed numeric nutrient criteria for this estuary. For more information on EPA's process of delineating, segmenting, and deriving numeric nutrient criteria for this estuary, please see Section 1.3.
[36] Section 62-302.700, F.A.C.

mean annual temperature of 68 °F (20 °C) (ANERR 2008). July through September are generally the wettest months, receiving an average of 7.1–7.3 in (18.0–18.5 cm) per month; November and December are driest, receiving an average of 3.5–3.6 in (8.9–9.1 cm) per month (based on a 30-year average). During the summer and fall, extreme weather events, particularly hurricanes, can also affect the local climate (ANERR 2008).

The streams in the Apalachicola watershed have been modified by dredge-and-fill activities related to logging, agricultural practices, and accommodating commercial river traffic (Boning 2007; FDEP 2005). In the Apalachicola watershed, the dominant land cover is pine plantations; wetlands are the second largest category (FDEP 2005). In the broader Apalachicola-Chattahoochee-Flint River Basin, developed areas account for 5 percent of land use; 29 percent is agriculture, 58 percent is forest, 5 percent is forested wetland, and 3 percent is water (Frick et al. 1998). FDEP (2005) similarly reported that disturbed community types (e.g., intensively managed grass areas, agricultural lands, exotic plant communities, and developed areas) cover more than 32 percent of the Apalachicola Basin, with the remaining 68 percent covered by natural communities (e.g., wetlands, forest, open water). A considerable amount of land in the Apalachicola Basin is in conservation (NWFWMD 2007).

The Apalachicola River, which is an alluvial river, provides the largest volume of freshwater and nutrient loads to the bay and estuary system, and plays an integral role in the salinity regime and ecology of the bay (ANERR 2008; FDEP 2010). The Apalachicola River features floodplain lakes and is characterized by annual/seasonal flooding, heavy sediment loads, and variable flow (NWFWMD 2007). Although the main tributary of the Apalachicola River is the Chipola River, the majority (more than 80%) of Apalachicola River flow comes from the upstream Chattahoochee and Flint rivers (ANERR 2008; Mattraw and Elder 1984). Of the six major aquifers underlying the basin, only the surficial and Floridan are in the Florida portion of the Apalachicola-Chattahoochee-Flint Basin and affect the reserve (ANERR 2008).

2.6.2.2. Impaired Waters[37]

No Class II or Class III marine WBIDs are in the Apalachicola Bay area that have been listed for a nutrient-related parameter on Florida's CWA section 303(d) list approved by EPA.[38] No Class II or Class III marine WBIDs with nutrient-related TMDLs were documented for this region.[39]

[37] For more information about the data source, see Volume 1, Appendix A.

[38] The nutrient-related 303(d) list was a compilation of EPA-approved 303(d) listing information for nutrients, chl-a, and DO provided in three decision documents: September 2, 2009, Basin Groups 1, 2, and 5 EPA Decision Document (http://www.epa.gov/region4/water/tmdl/florida/documents/fl09303d_decisiondoc_090209.pdf);
May 13, 2010, Basin Group 3 EPA Decision Document
(http://www.epa.gov/region4/water/tmdl/florida/documents/fl303d_%20partialapproval_decision_docs051410.pdf);
and the December 21, 2010, Basin Group 4 EPA Decision Document
(http://www.epa.gov/region4/water/tmdl/florida/documents/group_4_final_dec_doc_and_partial_app_letter_12_21_10.pdf).

[39] TMDLs were identified in February 2011 by compiling nutrient-related draft/final TMDLs from the following three sources:
FDEP TMDL website (http://www.dep.state.fl.us/water/tmdl/index.htm)
EPA Region 4 website (http://www.epa.gov/region4/water/tmdl/florida/index.html)
EPA National WATERS expert query tool (http://www.epa.gov/waters/tmdl/expert_query.html).

2.6.2.3. Water Quality

Water quality characteristics of Apalachicola Bay are primarily determined by a combination of flow from the Apalachicola River, local runoff, and Gulf of Mexico coastal waters (ANERR 2008). Upstream, out-of-state land and water uses also affect river health (Livingston 2008). Alabama and Georgia (especially the greater Atlanta metropolitan area) withdraw large quantities from the river system and underground aquifers on a daily basis (NWFWMD 2007). During low-flow cycles, those demands can have significant effects on the bay (Livingston 2008; NWFWMD 2007). Nonpoint source pollution includes animal manure (primarily chicken litter); runoff from agricultural, urban, and suburban areas; septic systems; decomposition of organic matter; and atmospheric deposition (Frick et al. 1998). The Apalachicola River provides the main input of nutrients to Apalachicola Bay and Estuary (ANERR 2008; FDEP 2010; Leitman and Howell 1991).

River flow, local runoff, tidal interaction, residence time, resuspension of sediments, and flux from benthic sediments influence nutrient concentrations in the bay. Turbidity parallels freshwater inflow regimes to the bay, but it is also affected by wind events, which tend to resuspend bottom sediments. Higher turbidity is generally observed during periods of high river flow or storm events, particularly hurricanes (ANERR 2008). During non-storm conditions measured before Hurricane Frances (2004), mean and maximum concentrations of total suspended solids (TSS) were 26.8 and 58 mg/L, respectively (Chen et al. 2009). Although water color measurements in Apalachicola Bay can range from 0 to more than 300 platinum-cobalt units (PCU), typical measurements are in the range of 20 to 160 PCU. Lower values are generally measured near the Gulf of Mexico, and higher values generally occur with local rainfall in the upper reaches of East Bay and at the mouth of the river during high river flow conditions (ANERR 2008).

Livingston (1984) found that typically, peak concentrations of DO occurred during winter and spring; lower DO concentrations occurred in the summer and fall. Vertical stratification and differences in DO concentrations existed in different parts of the system (Livingston 1984). Low DO levels (< 4 mg/L) have been measured in some portions of the bay. Seasonal and diurnal variations in DO are pronounced in the upper reaches of the bay, particularly during the summer (ANERR 2008).

Salinities in Apalachicola Bay also followed a seasonal trend between 1972 and 1982, with values that were lowest during the winter and spring (when river flow was highest) and highest during the fall drought period of October and November (when river flow was at a minimum) (Livingston 1984). Spatially, salinity values generally increase with southward distance from the river mouth, but throughout the year, salinity values typically range from 0 to 33 PSU and are attributed to the dynamic nature of the bay (ANERR 2008).

Mortazavi et al. (2000c) reported an annual mean chl-a concentration of 11.84 µg/L for a 3-year period beginning in 1993. During the summer, they found low chl-a concentrations, high phytoplankton productivity, high zooplankton abundance, low river flow, and low nutrient input to the estuary. Annually, 80 percent of the chl-a loss from the estuary was attributed to zooplankton grazing in the water column (Mortazavi et al. 2000c).

The Apalachicola River is the main source of nitrogen in Apalachicola Bay and Estuary, providing approximately 83 percent of TN; the remainder comes from the St. George Sound area (ANERR 2008). Mortazavi et al. (2000a) found that the river provided 73 percent of the dissolved organic nitrogen (DON) and 92 percent of total DIN between June 1994 and May 1996, but they did not find a clear seasonal pattern in river concentrations of DIN and DON. The mean river DIN concentration (0.350 ± 0.021 mg nitrogen/L) was significantly higher than the mean river DON concentration (0.183 ± 0.020 mg nitrogen/L). At the estuary's eastern boundary, DIN and DON concentrations were 0.115 ± 0.020 and 0.208 ± 0.014 mg nitrogen/L, respectively (Mortazavi et al. 2000a).

Approximately 78 percent of TP input to the estuary comes from the Apalachicola River (Mortazavi et al. 2000b). Mortazavi and colleagues (2000b) reported that, on average, TP input to the estuary was 0.015 oz phosphorus/ft^2/yr (4.57 g phosphorus/m^2/yr), primarily in particulate form. Overall, for the Apalachicola River, Mortazavi et al. (2000b) reported no clear seasonal trends for and soluble reactive phosphorus, dissolved organic phosphorus, or particulate phosphorus.

2.6.2.4. Biological Characteristics

The Apalachicola Basin has been characterized as one of the most ecologically diverse and significant natural areas in the United States (Livingston 2008; NWFWMD 2007). The Apalachicola drainage basin is home to more than 1,300 species of plants, 50 mammalian species, 40 amphibian species, and 80 species of reptiles (Apalachicola Riverkeeper 2011). Habitat structure and biological processes in the estuarine system are driven by several factors including river flow, basin physiography, nutrient inputs, salinity, and wind (Livingston 1984). More recently, Livingston (2008) noted that declining water levels and reduced seasonal flooding, especially in conjunction with droughts, have resulted in adverse effects on biological communities.

Salt, brackish, and fresh marshes are distributed primarily in the inter-tidal areas along the perimeter of the bay and the delta area of the lower river and East Bay, covering almost 20 percent of the total aquatic area of the Apalachicola Bay system (ANERR 2008).

Submerged aquatic vegetation (SAV) habitats are generally found only in the lower sections of the Apalachicola River near the bay (ANERR 2008). Livingston (1984) reported that in Apalachicola Bay, approximately 70 percent of the open water was characterized as a subtidal, unvegetated, soft-sediment area, with the remaining 30 percent composed of SAV and oyster reefs. Marine, brackish, and freshwater species of SAV are found in the Apalachicola Bay system and provide habitat and organic matter important for many species (ANERR 2008; Livingston 2008).

In Apalachicola Bay, phytoplankton is generally the base of the marine food web and is a significant contributor to the bay's exceptional productivity (ANERR 2008). In a 2000 study, phytoplankton productivity varied considerably and peaked during the summer (Mortazavi et al. 2000c). The Apalachicola River and Bay support a substantial seafood harvest, including oyster, shrimp, and numerous finfish (Livingston 2008). More than 10 percent of the nation's oysters and 90 percent of Florida's oysters are harvested from the bay (Apalachicola Riverkeeper 2011).

Oyster bars provide habitat for a variety of estuarine organisms (Livingston 1984). A 2008 study showed that prolonged high salinity (such as due to droughts and reduced riverine inputs of freshwater) increased predation and disease in oysters (Livingston 2008). The Apalachicola Bay and River support an extensive commercial and recreational fishery and are home to several endemic species (ANERR 2008; NWFWMD 2007).

For a more detailed summary of this water body, see Volume 1, Appendix A.

2.6.3. Data Used

Several data sources specific to Apalachicola Bay were used in addition to those sources described in Section 1.4.3, as summarized in Table 2-46.

Table 2-46. Data sources specific to Apalachicola Bay models

Data	Source	Location Used
Springs discharge and water quality data	Florida Department of Environmental Protection (FDEP 2004, 2009)	Apalachicola, Suwannee, Econfina, Waccasassa, Crystal, and Withlacoochee watershed models
Municipal and industrial point sources	Alabama Department of Environmental Management (ADEM No date a, No date b)	Apalachicola watershed model
Water quality data	Alabama Department of Environmental Management (ADEM 2006)	Apalachicola watershed model
FDEP Level III Florida Land Use	Northwest Florida Water Management District (NWFWMD 2009)	Apalachicola and Apalachee watershed models
FDEP Level III Florida Land Use	Suwannee River Water Management District (SRWMD No date)	Apalachee, Econfina, Suwannee, and Waccasassa watershed models
Climate data	Florida State Climate Center (FSCC 2009) and EarthInfo (EarthInfo 2009)	All watershed models
Municipal and industrial point sources	Georgia Environmental Protection Division (GAEPD No date)	Suwannee and Apalachee watershed models
Water quality data	Georgia Environmental Protection Division (GAEPD 2008)	Suwannee and Apalachee watershed models
Georgia Land Use Trends land use	Natural Resources Spatial Analysis Lab (NARSAL 2008)	Suwannee and Apalachee watershed models
FDEP Level III Florida Land Use	Southwest Florida Water Management District (SWFWMD 2007)	Waccasassa, Crystal, and Withlacoochee watershed models

2.6.4. Segmentation

The isohaline GIS analysis and geomorphological structure of Apalachicola Bay yielded five distinct segments. The segments were limited to a distance of 3 nautical miles from the shoreline, where the estuary numeric nutrient criteria would be applicable. Figure 2-21 shows the resulting segments for Apalachicola Bay.

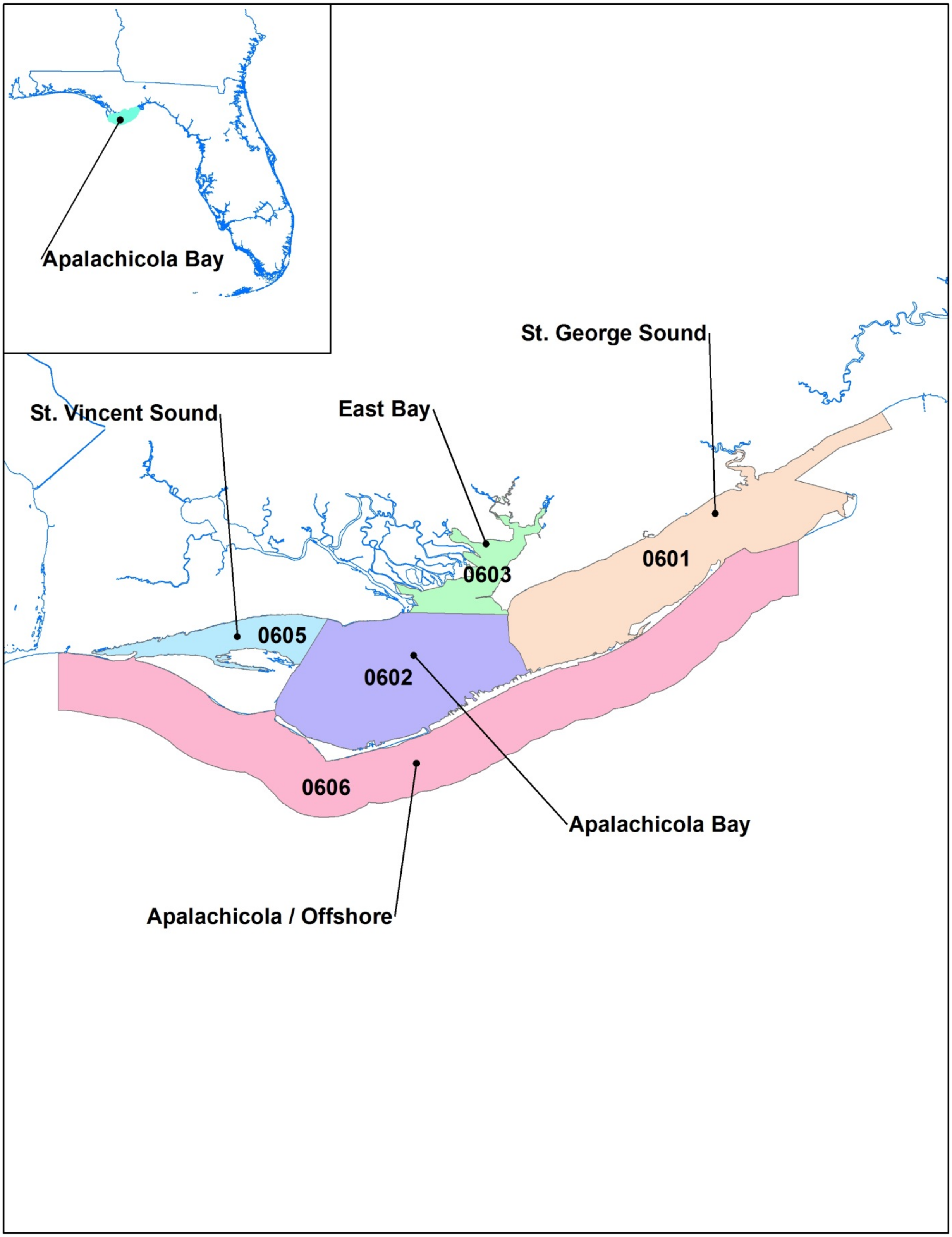

Figure 2-21. Results of Apalachicola Bay segmentation

2.6.5. Water Quality Targets

2.6.5.1. Seagrass Depth and Water Clarity Targets

Seagrass depth of colonization (measured as Z_c) and water clarity (measured as K_d) targets (Table 2-47) were established for three estuary segments based on the 1992 seagrass coverage for Apalachicola Bay (Figure 2-22) (USGS 2012). A minimal amount of seagrass was present within segment 0605 (Figure 2-22), but it is unlikely that it provides a representative indicator of water quality throughout the segment, therefore no depth of colonization or water clarity target was established for this segment. Evaluation of CDOM in the Apalachicola Bay segments showed that CDOM in segment 0602 was likely to attenuate photosynthetically active radiation to below 20 percent at 1.5 m, which is the estimate of depth of colonization in 1992. This could reflect the largely offshore distribution of seagrass within the segment (Figure 2-22). Based on this observation, the depth of colonization target was reduced to 1.0 m, allowing for the same photosynthetically active radiation level observed at the depth of colonization in the other segments (Table 2-47).

Table 2-47. Apalachicola Bay seagrass depth and water clarity targets by segment. Estimates of K_d(CDOM) were computed from color data in the IWR Run 40 database.

Estuary Segment	Depth of Colonization (Z_c) Target (m)	Light Attenuation Coefficient (K_d) Target (1/m)	K_d (CDOM)
0601	1.6	1.0	0.88
0602	0.9	1.8	1.45
0603	0.6	2.6	2.30
0605	No target	-	-
0606	No target	-	-

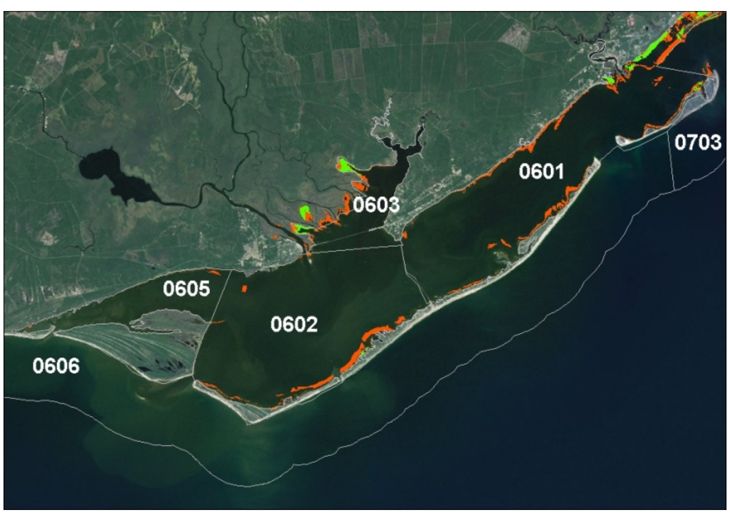

Figure 2-22. 1992 seagrass coverage for Apalachicola Bay

2.6.5.2. Chlorophyll a Target

To prevent nuisance algal blooms and protect the estuary's designated uses, chl-a levels must not exceed 20 µg/L more than 10 percent of the time. The rationale for that target is provided in Section 1.2.2.

2.6.5.3. Dissolved Oxygen Targets

On the basis of the rationale that sufficient DO is necessary to protect aquatic life, as described in Section 1.2.3, the following DO targets were established:

- Minimum allowable DO of 4.0 mg/L as a water column average for each estuary segment 90 percent of the time over the simulation's time span
- Daily average DO of 5.0 mg/L as a water column average for each estuary segment 90 percent of the time over the simulation's time span
- Minimum 3-hour average DO of 1.5 mg/L in the bottom two layers for each estuary segment over the simulation's time span

2.6.6. Results of Analyses

2.6.6.1. Mechanistic Model Analysis

Average load contributions from the Apalachicola watershed are shown in Table 2-48.

Table 2-48. Average load contributions from the Apalachicola watershed (2002–2009)

	TN Load[1] (kg/d; Mean ± se)			TP Load[1] (kg/d; Mean ± se)		
Year	Existing[2]	Background[3]	Anthropogenic[4]	Existing[2]	Background[3]	Anthropogenic[4]
2002	3,191 ± 238	3,153 ± 236	38 ± 2	73 ± 6	68 ± 6	5 ± 0
2003	5,023 ± 303	4,970 ± 300	53 ± 3	141 ± 12	134 ± 11	6 ± 0
2004	1,714 ± 129	1,689 ± 128	25 ± 1	42 ± 4	37 ± 4	4 ± 0
2005	3,093 ± 261	3,057 ± 259	36 ± 2	77 ± 8	72 ± 8	5 ± 0
2006	772 ± 79	757 ± 78	16 ± 1	16 ± 1	13 ± 1	2 ± 0
2007	726 ± 113	712 ± 112	14 ± 1	14 ± 2	12 ± 2	2 ± 0
2008	1,008 ± 90	991 ± 89	17 ± 1	23 ± 3	21 ± 2	2 ± 0
2009	3,456 ± 310	3,417 ± 308	39 ± 3	78 ± 8	74 ± 8	4 ± 0

[1] Annual average daily load from all terminal reaches
[2] Annual average daily load from all terminal reaches computed from existing model scenario
[3] Annual average daily load from all terminal reaches computed from background model scenario (remove anthropogenic sources/land use; use existing hydrology)
[4] Anthropogenic load computed as difference between existing and background load

For Apalachicola Bay, DO, chl-a, and light attenuation coefficient targets were met for all segments using 2002–2009 nutrient loads. Table 2-49 identifies which targets were met under 2002–2009 nutrient loads and which targets were not met.

An evaluation of model sensitivity to the water quality targets applied revealed that all targets were sensitive to changes in nutrients in Apalachicola Bay.

Table 2-49. Water quality targets met for Apalachicola Bay based on mechanistic modeling

Segment	DO	Chl-a	K_d
0601	Yes	Yes	Yes
0602	Yes	Yes	Yes
0603	Yes	Yes	Yes
0605	Yes	Yes	Yes
0606	Yes	Yes	No target

A summary of candidate criteria based on 2002–2009 nutrient loads for Apalachicola Bay Estuary segments is given in Table 2-50.

Table 2-50. Summary of candidate criteria for Apalachicola Bay derived from mechanistic modeling

Segment	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
0601	0.53	0.019	3.6
0602	0.51	0.019	2.7
0603	0.76	0.034	1.7
0605	0.52	0.016	11.9
0606	0.30	0.008	2.3

2.6.6.2. Statistical Model Analysis

Data were not sufficient within each segment to conduct statistical analyses in Apalachicola Bay.

2.6.7. Application of Analyses for Proposed Numeric Nutrient Criteria

Data necessary to conduct statistical analyses were not available for any of the segments in Apalachicola Bay. However, the mechanistic model provided values for every segment in the estuary. EPA derived the proposed numeric nutrient criteria for Apalachicola Bay shown in the table below based on the mechanistic modeling results.

EPA found that all three endpoints were achieved under the calibrated 2002–2009 nutrient loads. EPA also found that all three endpoints were sensitive to changes in nutrients. The proposed criteria were derived to be protective of all three. The values under mechanistic modeling represent the 90th percentile annual geometric mean nutrient concentrations from the 2002–2009 modeled nutrient loads.

Proposed numeric nutrient criteria for TN, TP, and chl-a in Apalachicola Bay segments are summarized in Table 2-51.

Table 2-51. Proposed and candidate numeric nutrient criteria for Apalachicola Bay segments

		Proposed Criteria			Mechanistic Modeling		
SEGMENT	SEGMENT ID	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
St. George Sound	0601	0.53	0.019	3.6	0.53	0.019	3.6
Apalachicola Bay	0602	0.51	0.019	2.7	0.51	0.019	2.7
East Bay	0603	0.76	0.034	1.7	0.76	0.034	1.7
St. Vincent Sound	0605	0.52	0.016	11.9	0.52	0.016	11.9
Apalachicola/Offshore	0606	0.30	0.008	2.3	0.30	0.008	2.3

2.6.8. Downstream Protective Values

In Apalachicola Bay mechanistic models were applied to derive proposed DPVs for TN and TP shown in Table 2-52.

Table 2-52. Proposed DPVs for Apalachicola Bay

Tributary[a]	LSPC Model Watershed ID	USGS Reach Code	Estuary Segment	TN (mg/L)	TP (mg/L)
Depot Creek	50025	03130011001132	0605	1.17	0.017
	50026	03130011004589	0605	0.76	0.020
Huckleberry Creek	50027	03130011000427	0605	0.99	0.014
Graham Creek	50031	03130011000895	0603	0.85	0.014
Whiskey George Creek	50075	03130013000092	0603	1.12	0.018
Bear Creek	50077	03130013000219	0603	1.42	0.018
Salt water	50078	03130013000189	0603	1.12	0.018
Miller Creek	50080	03130013000541	0601	1.44	0.024
	50096	03130011004591	0605	0.60	0.011
	50098	03130011004584	0605	1.28	0.018
East River	50113	03130011001113	0602	0.85	0.042
Cash Creek	50114	03130013000179	0603	1.19	0.018
Apalachicola River	50115	03130011001114	0603	1.06	0.020
Doyle Creek	50116	03130013000089	0603	1.22	0.018
Carrabelle River	50117	03130013000007	0601	0.74	0.024

[a] Tributary names left blank are unnamed

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2.7. Alligator Harbor

2.7.1. Proposed Numeric Nutrient Criteria

Proposed numeric nutrient criteria for TN, TP, and chl-a in Alligator Harbor segments are summarized in Table 2-53.

Table 2-53. Proposed numeric nutrient criteria for Alligator Harbor segments

Segment Name	Segment Number	TN mg/L)	TP (mg/L)	Chl-a (µg/L)
Alligator Harbor	0701	0.36	0.011	2.8
Alligator/Offshore	0702	0.33	0.009	3.1
Alligator/Offshore	0703	0.33	0.009	2.9

2.7.2. General Characteristics

2.7.2.1. System Description

Alligator Harbor is a shallow, high-salinity lagoon on the southeast coast of Franklin County in northwest Florida.[40] The basin has a small subwatershed with no major freshwater riverine inputs, but it has regular tidal exchange with the Gulf of Mexico (FDEP 2010). The 6.3 mi^2 (16 km^2 [4,045 ac]) harbor lies entirely in the Alligator Harbor Aquatic Preserve, which covers nearly 22 mi^2 (57 km^2 [14,184 ac]) of submerged lands (FDEP 2010; FDNR 1986; FDEP 2011). Alligator Harbor is designated as an EPA Gulf of Mexico Ecological Management Site (FDEP 2010), and Alligator Harbor Aquatic Preserve is designated as an OFW.[41]

[40] The information presented in this system description was compiled to summarize local information pertaining to the proposed numeric nutrient criteria for this estuary. For more information on EPA's process of delineating, segmenting, and deriving numeric nutrient criteria for this estuary, please see Section 1.3.
[41] Section 62-302.700, F.A.C.

Alligator Harbor is between the climates of subtropical peninsular Florida and temperate southeastern United States. Average rainfall is about 57 in (145 cm) (FDNR 1986). Based on a 30-year average, July to September are generally the wettest months (monthly average: 7.1–7.3 in [18.0–18.5 cm]); April and May are driest (monthly average: 2.6–3.0 in [6.6–7.6 cm]) (NWFWMD 2011). The harbor is affected by extreme weather events, such as hurricanes, during summer and fall (ANERR 2008). Land use consists of forested (58%), wetlands (22%), urban (14%), brushlands (2.3%), beaches (2.2%), and utilities (0.1%) (FDEP 2010).

The harbor is partially isolated from the Gulf of Mexico by a shallow sandbar at Peninsula Point; a deep channel at the northern end of the sandbar connects the bay to the Gulf. Alligator Harbor is generally shallow (mean low water depth of about 13 ft [4 m]) (FDEP 2010; FDNR 1986; Schmidt 1978). A lack of riverine input and sandy sediments help produce generally clear waters (FDEP 2010). Because the harbor has no significant freshwater inputs and has regular tidal exchange with the Gulf of Mexico, salinity levels are relatively stable and basically the same as the Gulf of Mexico (FDNR 1986). Only during heavy rains does substantial freshwater enter the harbor from runoff and significantly lower salinity in the harbor (FDEP 2010; FDNR 1986).

2.7.2.2. Impaired Waters[42]

No water body WBIDs with a nutrient-related parameter are on Florida's CWA section 303(d) list approved by EPA in the Alligator Harbor area.[43] No Class II or Class III marine WBIDs with nutrient-related TMDLs were documented for this region.[44]

2.7.2.3. Water Quality

Nutrients in stormwater are considered the primary water quality threat to Alligator Harbor (FDEP 2009a). Other reported sources of nutrients include 118 septic tanks within 100 ft (30.5 m) of the shoreline. DO concentrations vary seasonally from 40 to 100 percent saturation. Turbidity values typically range from 0 to 264 NTU, with higher values usually associated with storm events (FDEP 2010). Chl-a, TN, and TP concentrations decreased between 2001 and 2008 (FDEP 2009a). The overall geometric mean for chl-a, TN, and TP during that period were 1.58 µg/L, 5.9 mg/L, and 3.31 mg/L, respectively. Across Alligator Harbor, salinity values were relatively stable and ranged from 28 to 38 PSU, with an average of 34 PSU (FDEP 2010). No further information was found in the available literature.

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[42] For more information about the data source, see Volume 1, Appendix A.

[43] The nutrient-related 303(d) list was a compilation of EPA-approved 303(d) listing information for nutrients, chl-a, and DO provided in three decision documents: September 2, 2009, Basin Groups 1, 2, and 5 EPA Decision Document (http://www.epa.gov/region4/water/tmdl/florida/documents/fl09303d_decisiondoc_090209.pdf);
May 13, 2010, Basin Group 3 EPA Decision Document
(http://www.epa.gov/region4/water/tmdl/florida/documents/fl303d_%20partialapproval_decision_docs051410.pdf);
and the December 21, 2010, Basin Group 4 EPA Decision Document
(http://www.epa.gov/region4/water/tmdl/florida/documents/group_4_final_dec_doc_and_partial_app_letter_12_21_10.pdf).

[44] TMDLs were identified in February 2011 by compiling nutrient-related draft/final TMDLs from the following three sources:
FDEP TMDL website (http://www.dep.state.fl.us/water/tmdl/index.htm)
EPA Region 4 website (http://www.epa.gov/region4/water/tmdl/florida/index.html)
EPA National WATERS expert query tool (http://www.epa.gov/waters/tmdl/expert_query.html).

2.7.2.4. Biological Characteristics

Major Alligator Harbor habitats and plant communities include marine SAV (especially seagrasses), salt marshes (wetlands), oyster reefs/bars, subtidal softbottom (unvegetated) substrates, algae, and beaches (FDEP 2010, 2011; FDNR 1986). The harbor contains extensive salt marsh habitat (FDNR 1986) composed primarily of smooth cordgrass (*Spartina alterniflora*) and black needlerush (*Juncus roemerianus*). At higher elevations, glasswort (*Salicornia virginica*), saltwort (*Batis maritime*), sea ox-eye (*Borrichia frutenscens*), and marsh elder (*Iva frutescens*) are common (FDEP 2010). Animal life is rich and diverse and includes primary consumers that feed on vascular plant detritus and fresh algae (FDNR 1986). Some mammals nest in the marsh; others come to feed during low tides (FDNR 1986).

Alligator Harbor supports one of the densest stocks of seagrass beds in northwest Florida (FDEP 2010). However, color, high turbidity, and sedimentation generally limit SAV growth in coastal areas of Franklin County, making grassbeds and associated algae more common in shallower portions of the estuary (FDNR 1986). In 1992 approximately 1.2 mi^2 (3.1 km^2 [755 ac]) of seagrasses and associated epiphyte coverage were measured in Alligator Harbor and associated shoals (FFWCC 2011).

Common types of macroalgae in Alligator Harbor, in order from most common to least, include red, green, brown, and blue-green algae. In general, algal growth in the harbor is rapid and continuous throughout the year (FDNR 1986). The softbottom substrates in Alligator Harbor are generally dominated by polychaete and amphipod invertebrates (Livingston 1984). Oyster reefs form islands of stable substrate in an otherwise muddy environment, providing essential habitat for many animals, especially sessile, suspension-feeding invertebrates such as barnacles and polychaetes, and mobile invertebrates such as crabs. Alligator Harbor's consistently high salinity and high summer temperatures limit oyster reef development below the intertidal zone. High salinities also allow oyster predation by organisms accustomed to consistently saline subtidal waters. For those reasons, and because the oysters are typically small, they are not harvested commercially (FDNR 1986).

Alligator Harbor supports a wide variety of fish species, many of which are of commercial and recreational importance, including spotted seatrout, redfish, and flounder (FDEP 2010; FDEP 2011; FDNR 1986). Sporadic fish kills potentially have been attributed to unusually high freshwater runoff into the harbor (FDEP 2010).

For a more detailed summary of this water body, see Volume 1, Appendix A.

2.7.3. Data Used

Several data sources specific to Alligator Harbor were used in addition to those sources described in Section 1.4.3; those are summarized in Table 2-54.

Table 2-54. Data sources specific to Alligator Harbor models

Data	Source	Location Used
Springs discharge and water quality data	Florida Department of Environmental Protection (FDEP 2004, 2009b)	Apalachicola, Suwannee, Econfina, Waccasassa, Crystal, and Withlacoochee watershed models
FDEP Level III Florida Land Use	Suwannee River Water Management District (SRWMD No Date)	Apalachee, Econfina, Suwannee, and Waccasassa watershed models
Climate data	Florida State Climate Center (FSCC 2009) and EarthInfo (EarthInfo 2009)	All watershed models
Municipal and industrial point sources	Georgia Environmental Protection Division (GAEPD No date)	Suwannee and Apalachee watershed models
Water quality data	Georgia Environmental Protection Division (GAEPD 2008)	Suwannee and Apalachee watershed models
Georgia Land Use Trends land use	Natural Resources Spatial Analysis Lab (NARSAL 2008)	Suwannee and Apalachee watershed models
FDEP Level III Florida Land Use	Southwest Florida Water Management District (SWFWMD 2007)	Waccasassa, Crystal, and Withlacoochee watershed models

2.7.4. Segmentation

Alligator Harbor was segmented on the basis of existing WBIDs. The WBIDs were defined up to a distance of 3 nautical miles and represented the seagrass distribution in the estuary. Figure 2-23 shows the resulting segments for Alligator Harbor.

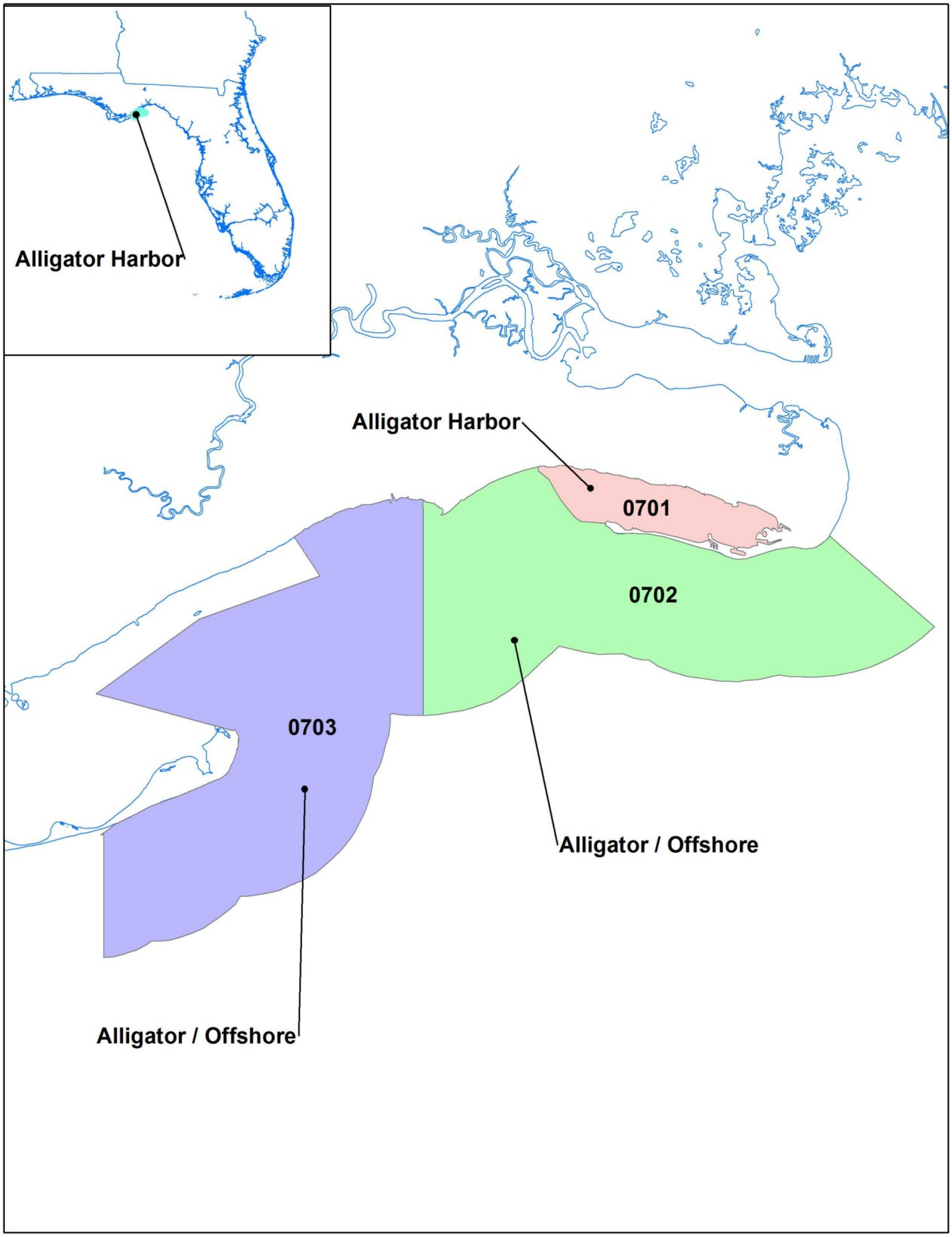

Figure 2-23. Results of Alligator Harbor segmentation

2.7.5. Water Quality Targets

2.7.5.1. Seagrass Depth and Water Clarity Targets

Seagrass coverage data from 1992 were considered for Alligator Harbor (Figure 2-24) (FWRI 2012). However, seagrass depth of colonization and water clarity targets were not established because bathymetric data needed for the calculation could not be located.

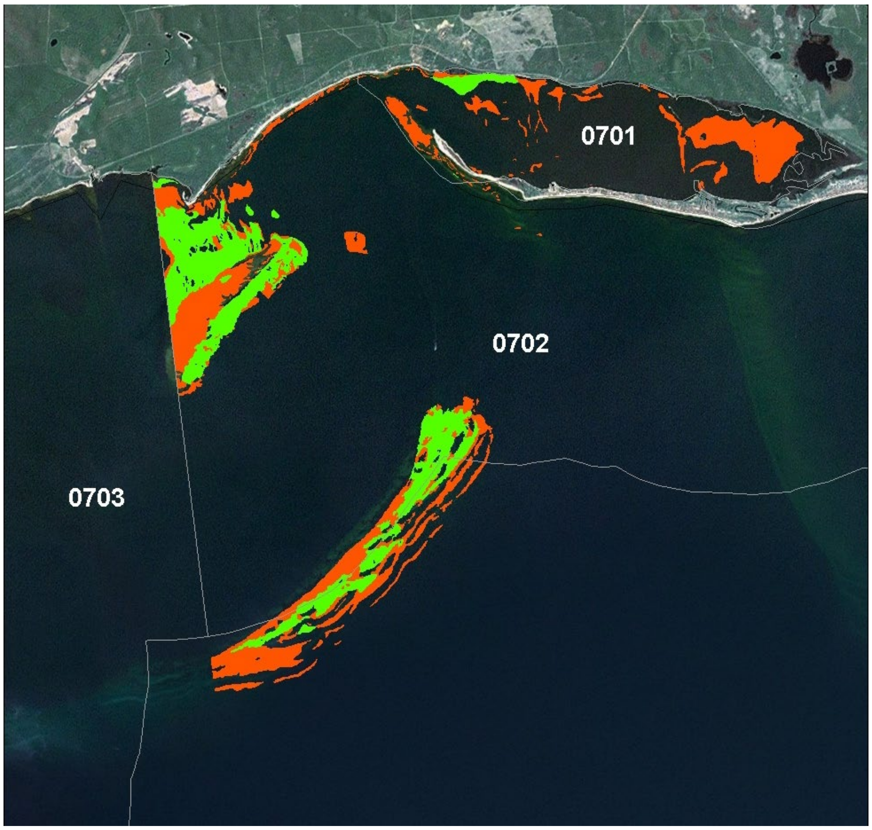

Figure 2-24. Seagrass coverage in the vicinity of Alligator Harbor in 1992. A seagrass area was associated with an offshore sand bar approximately 3 nautical miles from the coast. The apparent western limit of seagrass at the boundary of segments 0702 and 0703 likely results from the spatial limits of the seagrass coverage data, rather than the seagrass itself.

2.7.5.2. Chlorophyll a *Target*

To prevent nuisance algal blooms and protect the estuary's designated uses, chl-a levels must not exceed 20 µg/L more than 10 percent of the time. The rationale for that target is provided in Section 1.2.2.

2.7.5.3. Dissolved Oxygen Targets

On the basis of the rationale that sufficient DO is necessary to protect aquatic life, as described in Section 1.2.3, the following DO targets were established:

- Minimum allowable DO of 4.0 mg/L as a water column average for each estuary segment 90 percent of the time over the simulation's time span

- Daily average DO of 5.0 mg/L as a water column average for each estuary segment 90 percent of the time over the simulation's time span
- Minimum 3-hour average DO of 1.5 mg/L in the bottom two layers for each estuary segment over the simulation's time span

2.7.6. Results of Analyses

2.7.6.1. Mechanistic Model Analysis

Average load contributions to Alligator Harbor from the Apalachee watershed are shown in Table 2-55.

Table 2-55. Average load contributions to Alligator Harbor from the Apalachee watershed (2002–2009)

	TN Load[1] (kg/d; Mean ± se)			TP Load[1] (kg/d; Mean ± se)		
Year	Existing[2]	Background[3]	Anthropogenic[4]	Existing[2]	Background[3]	Anthropogenic[4]
2002	40 ± 2	37 ± 2	4 ± 0	3 ± 0	2 ± 0	1 ± 0
2003	73 ± 4	66 ± 4	7 ± 1	4 ± 0	3 ± 0	1 ± 0
2004	43 ± 2	39 ± 2	4 ± 0	3 ± 0	2 ± 0	1 ± 0
2005	65 ± 5	60 ± 4	5 ± 1	4 ± 0	3 ± 0	1 ± 0
2006	25 ± 2	22 ± 2	2 ± 0	2 ± 0	1 ± 0	1 ± 0
2007	12 ± 2	10 ± 1	2 ± 1	1 ± 0	1 ± 0	0 ± 0
2008	17 ± 2	15 ± 1	3 ± 0	2 ± 0	1 ± 0	1 ± 0
2009	58 ± 6	53 ± 5	5 ± 1	4 ± 0	3 ± 0	1 ± 0

[1] Annual average daily load from all terminal reaches
[2] Annual average daily load from all terminal reaches computed from existing model scenario
[3] Annual average daily load from all terminal reaches computed from background model scenario (remove anthropogenic sources/land use; use existing hydrology)
[4] Anthropogenic load computed as difference between existing and background load

For Alligator Harbor, DO and chl-a targets were met for all segments using 2002–2009 nutrient loads. Table 2-56 identifies which targets were met under 2002–2009 nutrient loads and which targets were not met.

An evaluation of model sensitivity to the biological endpoints applied also revealed that all water quality targets were sensitive to changes in nutrients in Alligator Harbor.

Table 2-56. Water quality targets met for Alligator Harbor based on mechanistic modeling

Segment	DO	Chl-a	K_d
0701	Yes	Yes	No target
0702	Yes	Yes	No target
0703	Yes	Yes	No target

A summary of candidate criteria based on 2002–2009 nutrient loads for Alligator Harbor Estuary segments is given in Table 2-57.

Table 2-57. Summary of candidate criteria for Alligator Harbor derived from mechanistic modeling

Segment	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
0701	0.36	0.011	2.8
0702	0.33	0.009	3.1
0703	0.33	0.009	2.9

2.7.6.2. Statistical Model Analysis

Data were not sufficient within each segment to conduct statistical analyses in Alligator Harbor.

2.7.7. Application of Analyses for Proposed Numeric Nutrient Criteria

Data necessary to conduct statistical analyses were not available for all segments of Alligator Harbor/Alligator Offshore. However, the mechanistic model provided values for every segment in the estuary. As a result, EPA derived the proposed numeric nutrient criteria for Alligator Harbor/Alligator Offshore shown in the table below based on the mechanistic modeling results.

Because depth of colonization targets were not available in Alligator Harbor/Alligator Offshore, EPA evaluated two endpoints in the mechanistic modeling approach: (1) chl-a concentrations associated with balanced phytoplankton biomass and (2) DO concentrations sufficient to maintain aquatic life. EPA found that both of the endpoints were met under the calibrated 2002–2009 nutrient loads and sensitive to nutrient changes. The proposed criteria were derived to be protective of both. The values under mechanistic modeling represent the 90th percentile annual geometric mean nutrient concentrations from the 2002–2009 modeled nutrient loads.

Proposed numeric nutrient criteria for TN, TP, and chl-a in Alligator Harbor segments are summarized in Table 2-58.

Table 2-58. Proposed and candidate numeric nutrient criteria for Alligator Harbor segments

		Proposed Criteria			Mechanistic Modeling		
SEGMENT	SEGMENT ID	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
Alligator Harbor	0701	0.36	0.011	2.8	0.36	0.011	2.8
Alligator/Offshore	0702	0.33	0.009	3.1	0.33	0.009	3.1
Alligator/Offshore	0703	0.33	0.009	2.9	0.33	0.009	2.9

2.7.8. Downstream Protective Values

In Alligator Harbor mechanistic models were applied to derive proposed DPVs for TN and TP shown in Table 2-59.

Table 2-59. Proposed DPVs for Alligator Harbor

Tributary[a]	LSPC Model Watershed ID	USGS Reach Code	Estuary Segment	TN (mg/L)	TP (mg/L)
	60001	03130013000559	0702	0.59	0.032

[a] Tributary names left blank are unnamed

2.7.9. References

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2.8. Ochlockonee Bay

2.8.1. Proposed Numeric Nutrient Criteria

Proposed numeric nutrient criteria for TN, TP, and chl-a in Ochlockonee Bay segments are summarized in Table 2-60.

Table 2-60. Proposed numeric nutrient criteria for Ochlockonee Bay segments

Segment Name	Segment Number	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
Ochlockonee-St. Marks Offshore	0825	0.79	0.033	2.7
St. Marks River Offshore	0827	0.51	0.022	1.7
St. Marks River	0828	0.55	0.030	1.2
Ochlockonee Offshore	0829	0.47	0.019	1.9
Ochlockonee Bay	0830	0.66	0.037	1.8

2.8.2. General Characteristics

2.8.2.1. System Description

Ochlockonee Bay is a small coastal plain estuary at the mouth of the Ochlockonee River in the northwest panhandle of Florida and is part of the greater Apalachee Bay watershed (FDEP 2010a; Seitzinger 1987).[45] The Ochlockonee Bay River Basin encompasses parts of Franklin, Wakulla, Liberty, Leon, and Gadsden counties and extends to Georgia (FDEP 2010a; Seitzinger 1987). The greater Apalachee Bay watershed consists of three basins (subwatersheds): the Ochlockonee River Basin, the St. Marks River Basin, and the Aucilla River Basin. The St. Marks River Basin, also part of the greater Apalachee Bay watershed, will be discussed in this system description. The lower part of the Ochlockonee River Basin contains the Apalachicola National Forest and the St. Marks National Wildlife Refuge (FDEP 2010a; Seitzinger 1987). On its western end, the bay receives freshwater inputs primarily from the Ochlockonee and Sopchoppy rivers. On its eastern end, the bay opens up to Apalachee Bay (Lazarevich 2007). The Northwest Florida Water Management District (NWFWMD) initially designated the Ochlockonee River and Ochlockonee Bay as Surface Water Improvement and Management (SWIM) Program priority water bodies for preservation and protection in 1988 (Lewis et al. 2009). Both water bodies remained on the updated 1996 SWIM priority list (NWFWMD 2006). FDEP has designated several water bodies in the St. Marks and Wakulla rivers as OFWs.[46]

The main land use in the Florida portion of the Ochlockonee River Basin is forestry. Land uses in the South Ochlockonee River Planning Unit primarily consist of upland forest (51%) and wetlands (43%) with limited urban and built-up (0.9%). In the Sopchoppy River Planning Unit, land uses consist of wetlands (51.7%) and upland forest (44.5%), with limited urban/built-up (0.8%). More than one-third of the basin is protected as publicly owned conservation and

[45] The information presented in this system description was compiled to summarize local information pertaining to the proposed numeric nutrient criteria for this estuary. For more information on EPA's process of delineating, segmenting, and deriving numeric nutrient criteria for this estuary, please see Section 1.3.
[46] Section 62-302.700, F.A.C.

management forest lands. Parts of the bay's shoreline are also preserved by local, state, and federal entities, including Bald Point State Park (FDEP 2006, 2010a). The urban/residential areas of the Ochlockonee River Basin predominantly adjoin the cities of Tallahassee and Quincy. Overall, the basin has a relatively low population density compared to other areas in the state (FDEP 2003, 2010a). Ochlockonee Bay is adjacent to two planning units: South Ochlockonee River Planning Unit and Sopchoppy River Planning Unit (FDEP 2010a).

Originating in Worth County, Georgia, the Ochlockonee River is a major freshwater contributor to Ochlockonee Bay. The Ochlockonee River Basin, which includes Ochlockonee Bay, covers an area of 2,416 mi^2 (6,257 km^2), with approximately 1,080 mi^2 (2,800 km^2) in Florida (FDEP 2011; NWFWMD 2006). The 206-mi (332-km) Ochlockonee River is fed by several tributaries including Telogia Creek, Little River, and the Sopchoppy River (FDEP 2003; NWFWMD 2006). Ochlockonee Bay is approximately 6 mi (10 km) long by 1.5 mi (2.4 km) wide (Seitzinger 1987). The bay is rapidly flushed, well mixed, and shallow with an average water residence time of 10 days and an average depth of 3–7 ft (1–2 m) (Kaul and Froelich 1984; Lazarevich 2007).

Ochlockonee Bay is underlain by the phosphatic Hawthorn Formation (Kaul and Froelich 1984). The predominant rock types underlying Franklin and Wakulla counties are Pleistocene/Holocene era clay, mud, sand, and alluvium (USGS 2011). Franklin and Wakulla counties are in the northwest region of Florida, which has four major groundwater systems. The Floridan and the surficial aquifer systems supply most of the groundwater in the area. More than 80 percent of the potable water supply in northwest Florida comes from groundwater supplies (NWFWMD No date). In the St. Marks River Basin, a large percentage of nonpoint source pollution from stormwater runoff enters the Floridan aquifer via sinks and swallets, which are characteristic of the karst topography (Lewis et al. 2009).

2.8.2.2. Impaired Waters[47]

No Class II or Class III marine WBIDs in the Ochlockonee Bay area are listed for a nutrient-related parameter on Florida's CWA section 303(d) list approved by EPA.[48] No Class II or Class III marine nutrient-related TMDLs were documented for this region.[49]

2.8.2.3. Water Quality

Limited water quality data are provided in the available literature for Ochlockonee Bay, partially because there is no existing monitoring program (FDEP 2010a). A majority of the water quality

[47] For more information about the data source, see Volume 1, Appendix A.

[48] The nutrient-related 303(d) list was a compilation of EPA-approved 303(d) listing information for nutrients, chl-a, and DO provided in three decision documents: September 2, 2009, Basin Groups 1, 2, and 5 EPA Decision Document (http://www.epa.gov/region4/water/tmdl/florida/documents/fl09303d_decisiondoc_090209.pdf);
May 13, 2010, Basin Group 3 EPA Decision Document
(http://www.epa.gov/region4/water/tmdl/florida/documents/fl303d_%20partialapproval_decision_docs051410.pdf);
and the December 21, 2010, Basin Group 4 EPA Decision Document
(http://www.epa.gov/region4/water/tmdl/florida/documents/group_4_final_dec_doc_and_partial_app_letter_12_21_10.pdf).

[49] TMDLs were identified in February 2011 by compiling nutrient-related draft/final TMDLs from the following three sources:
FDEP TMDL website (http://www.dep.state.fl.us/water/tmdl/index.htm)
EPA Region 4 website (http://www.epa.gov/region4/water/tmdl/florida/index.html)
EPA National WATERS expert query tool (http://www.epa.gov/waters/tmdl/expert_query.html).

and hydrologic data for the bay comes from empirical studies of the bay that focus on nutrient cycling and model analyses.

Population increases, such as those in Leon and Wakulla counties, have also contributed to increased nutrient loading from the additional wastewater generated (Lewis et al. 2009). Sources of nutrients to Ochlockonee Bay include nonpoint source inputs from septic tanks, urban and agricultural stormwater runoff, natural inputs (such as plant leaf litter), and atmospheric deposition. Stormwater runoff mainly enters the bay via the Ochlockonee and Sopchoppy rivers and their tributaries (FDEP 2010a). Kaul and Froelich (1984) suggested that Ochlockonee Bay has a lower annual productivity and nutrient recycling efficiency compared to larger estuaries because of its small size and rapid washout rates. Most of the regenerated nutrient fluxes in the bay are flushed out to the ocean rather than recycled in the estuary (Kaul and Froelich 1984). Nutrient concentrations tend to decrease from the head of the bay where the Ochlockonee River enters the mouth of the bay where it discharges to Apalachee Bay (Seitzinger 1987).

Lewis et al. (2009) found that average surface DO concentrations were greater than 7 mg/L at all water quality stations in the St. Marks and Wakulla rivers and Apalachee Bay except the St. Marks at Newport site in the upper tidal reach, which had a mean value of 6 mg/L. With mean values greater than 8 mg/L, DO concentrations were higher in Apalachee Bay than in the St. Marks and Wakulla rivers. DO concentrations in the Apalachee Bay were also more variable than concentrations in the rivers (Lewis et al. 2009).

During a 14-month study in Ochlockonee Bay from 1980 to 1981, chlorophyll and diatom count profiles were highest in the upper region of Ochlockonee Bay, suggesting that primary producers were most abundant in low-salinity waters (Kaul and Froelich 1984). Results from six sites in Apalachee Bay from the 2001 LakeWatch Program showed that mean chl-a concentrations ranged from 1.33 to 2.90 μg/L (FDEP 2010b). Chl-a concentrations were generally low throughout most of the St. Marks/Apalachee Bay system, with locally high levels in the St. Marks and Wakulla rivers, particularly in the middle and lower tidal reaches (Lewis et al. 2009). During their 1980–1981 study, Kaul and Froelich (1984) found that internal nitrate+nitrite (NO_3+NO_2) removal exceeded internal nitrate+nitrite input into the bay; approximately 58 percent of the fluvial nitrate+nitrite flux entering Ochlockonee Bay was removed through biological uptake; about 70 percent of the fluvial flux into the bay was discharged out of the bay. Biological productivity in the bay removed nutrients in the nitrogen-phosphorus ratio ≥ 8:1. The majority of all fluvial reactive phosphate (PO_4) in Ochlockonee Bay entered the ocean, and approximately 81 percent of the dissolved-reactive phosphate fluvial flux was removed biologically within the bay (Kaul and Froelich 1984).

Results from the 2001 LakeWatch Program in northern Apalachee Bay showed that mean TN and TP concentrations in Apalachee Bay offshore of the mouth of St. Marks River ranged from 0.21 to 0.29 mg/L and 0.0109 to 0.0152 mg/L, respectively (FDEP 2010b). Lewis et al. (2009) found low nutrient concentrations (both nitrogen and phosphorus) throughout estuarine portions of the St. Marks/Apalachee Bay system, except in the upper tidal reach of the Wakulla River. Apalachee Bay generally had lower TN and TP concentrations than both the St. Marks and Wakulla rivers (Lewis et al. 2009).

In 1994, upper portions of the Ochlockonee River were characterized by elevated turbidity, which was attributed to agricultural runoff and point sources originating outside Florida. High turbidity was also noted just below the Georgia-Florida state line, where agricultural runoff in Georgia was named as the source of turbidity and siltation. Further downstream, turbidity levels improved with the help of Lake Talquin, which settles out some of the turbidity. In Ochlockonee Bay, median values of 10 historical observations made between 1975 and 1979 showed a turbidity level of 10.0 NTU, a Secchi depth of 0.7 ft (0.2 m), color of 179 PCU, and 25 mg/L for TSS (Hand et al. 1994). No additional information was found in the available literature.

2.8.2.4. Biological Characteristics

Ochlockonee Bay contains several shallow estuarine and marine marshes, tidal flats, and shoals in the upper portion of the bay. Limited biological data are available for Ochlockonee Bay (FDEP 2010a). The St. Marks River Basin contains a variety of diverse habitats that support a wide range of riverine and estuarine biological communities, including palustrine forests, fresh and brackish wetlands, salt marshes, and SAV (Lewis et al. 2009).

Between the St. Marks and Ochlockonee river basins, wetlands make up about 18 percent of the total area. Most of those wetlands occur in areas of river floodplains with poor soil drainage or poorly drained swamps as nearshore marshes (FDEP 2003). Portions of Bald Point State Park, along the southern shoreline of Ochlockonee Bay, are dominated by mesic flatwoods and estuarine tidal marsh natural communities. The tidal marshes provide habitats for marine organisms and function as nurseries for many species of pelagic and deep-water fish (FDEP 2006).

Ochlockonee Bay contains many species of seagrass, rushes, and sedges along the coasts and river mouths including black needlerush (*Juncus roemerianus*), smooth cordgrass (*Spartina alterniflora*), saltwort (*Batis maritime*), glasswort (*Salicornia virginica*), sea ox-eye (*Borrichia frutescens*), and marsh elder (*Iva frutescens*). Sawgrass (*Cladium jamaicense*) is a common species in the upper reaches of the river mouth where the tidal marshes start to mix with freshwater marshes and swamps. The species form extensive estuarine and marine tidal marsh communities within Ochlockonee Bay (FDEP 2006, 2010a). No historical or current records of tidal marsh loss exist (FDEP 2010a).

Most SAV communities are present at the mouth of Ochlockonee Bay and in adjacent Apalachee Bay and Alligator Harbor. No seagrass monitoring programs are established in Ochlockonee Bay (FFWCC 2011). In the St. Marks/Apalachee Bay system, seagrass beds are dominated by widgeon grass (*Ruppia maritima*) and shoal grass (*Halodule wrightii*) inshore, and shoal grass, turtle grass (*Thalassia testudinum*), and manatee grass (*Syringodium filiforme*) offshore. Star grass (*Halophila engelmanni*) can also be found in this system. In the St. Marks River mouth, seagrass coverage decreased from 50 mi^2 (131 km^2 [32,310 ac]) in 2001 to 49 mi^2 (128 km^2 [31,510 ac]) in 2006 (FFWCC 2011).

During Kaul and Froelich's 14-month study from 1980 to 1981, Ochlockonee Bay exhibited pronounced seasonal variations in total estuarine phytoplankton productivity, with productivity in the early spring through early summer about three times higher than in the fall and winter. That seasonal pattern was associated with fluctuations in dissolved reactive phosphorus (DRP) and nitrate+nitrite riverine concentrations (Kaul and Froelich 1984). On the basis of historical red tide status maps produced by the FWRI, Ochlockonee Bay did not experienced episodic harmful algal blooms (HABs, e.g., *K. brevis*) between 2001 and 2011, nor any other major HAB events (FDEP 2010a; FWRI 2011).

Very little information is available about macroalgal communities and blooms in Ochlockonee Bay area. However, FDEP (2010a) states that no historical or current reports of macroalgal blooms exist in the bay.

Invertebrates living in Ochlockonee Bay's tidal marsh communities include marsh snails, periwinkles, mud snails, fiddler crabs, marsh crabs, green crabs, isopods, and amphipods (FDEP 2010a). No further information was found in the available literature.

Abundance, biomass, and species composition of fishes in the tidal salt marshes of St. Marks River is highly dynamic because of temporal variations in feeding migrations, recruitment of juveniles, and movement of resident species (Lewis et al. 2009).

The West Indian manatee (*Trichechus manatus latirostris*) is seasonally common to the St. Marks and Wakulla rivers, using the area as a summer feeding ground (FDEP 2001).

For a more detailed summary of this water body, see Volume 1, Appendix A.

2.8.3. Data Used

Several data sources specific to Ochlockonee Bay were used in addition to those sources described in Section 1.4.3, as summarized in Table 2-61.

Table 2-61. Data sources specific to Ochlockonee Bay models

Data	Source	Location Used
Springs discharge and water quality data	Florida Department of Environmental Protection (FDEP 2004, 2009)	Apalachicola, Suwannee, Econfina, Waccasassa, Crystal, and Withlacoochee watershed models
FDEP Level III Florida Land Use	Suwannee River Water Management District (SRWMD No date)	Apalachee, Econfina, Suwannee, and Waccasassa watershed models
Climate data	Florida State Climate Center (FSCC 2009) and EarthInfo (EarthInfo 2009)	All watershed models
Municipal and industrial point sources	Georgia Environmental Protection Division (GAEPD No date)	Suwannee and Apalachee watershed models
Water quality data	Georgia Environmental Protection Division (GAEPD 2008)	Suwannee and Apalachee watershed models
Georgia Land Use Trends land use	Natural Resources Spatial Analysis Lab (NARSAL 2008)	Suwannee and Apalachee watershed models
FDEP Level III Florida Land Use	Southwest Florida Water Management District (SWFWMD 2007)	Waccasassa, Crystal, and Withlacoochee watershed models

2.8.4. Segmentation

Ochlockonee Bay was divided into five segments on the basis of the existing WBIDs. Two WBIDs, 0825 and 0826, were combined into one segment (0825). The segments were limited to a distance of 3 nautical miles from the shoreline, where the estuary numeric nutrient criteria would be applicable. Figure 2-25 shows the resulting segments for Ochlockonee Bay.

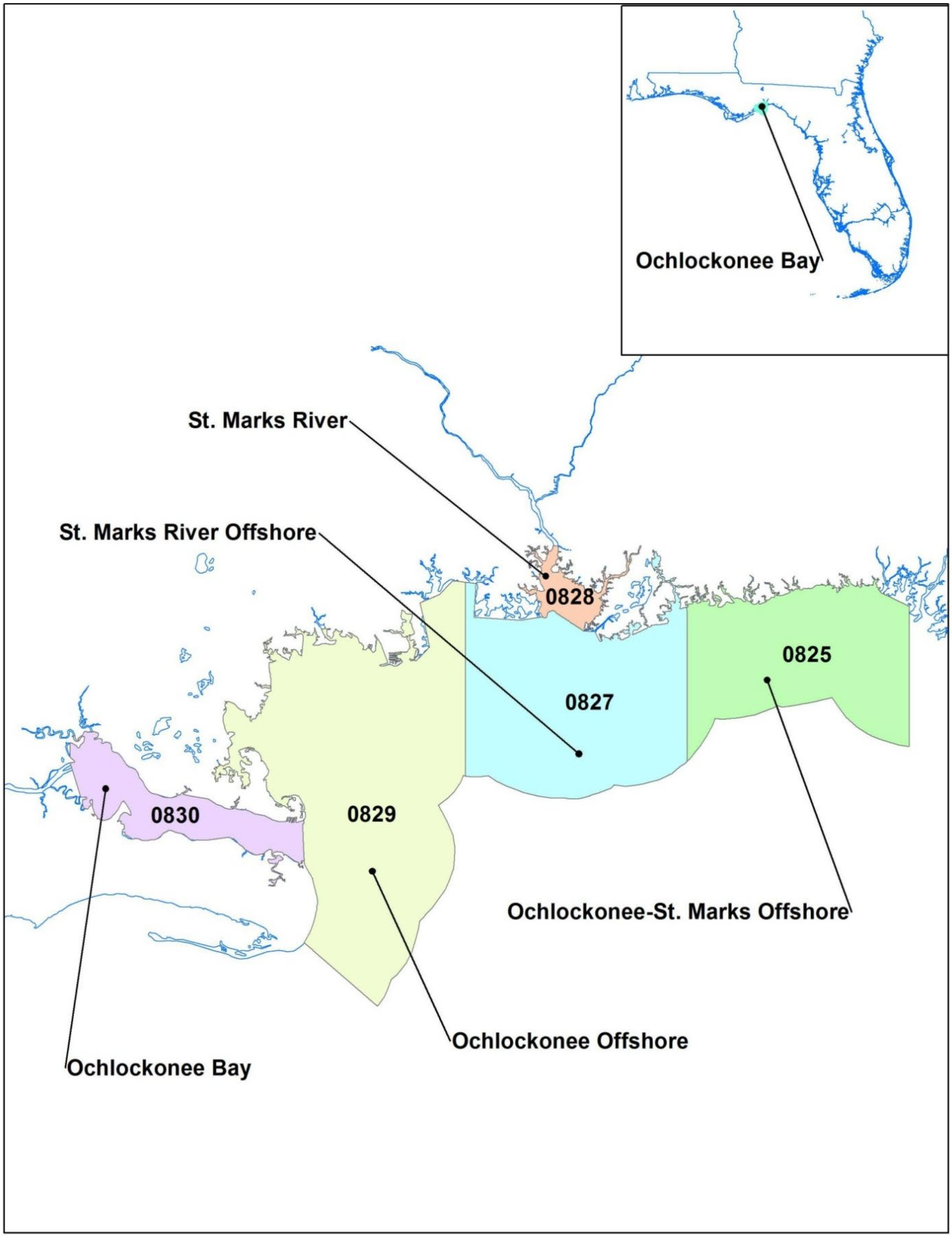

Figure 2-25. Results of Ochlockonee Bay segmentation

2.8.5. Water Quality Targets

2.8.5.1. Seagrass Depth and Water Clarity Targets

Seagrass depth of colonization (measured as Z_c) and water clarity (measured as K_d) targets were established for Ochlockonee Bay segments (Table 2-62) based on coverage data from 1992 (Figure 2-26) (FWRI 2012). Seagrasses were delineated in segment 0829, offshore from Ochlockonee Bay, but not in segment 0830. Seagrass coverage was very extensive in the vicinity of St. Marks River, particularly in offshore segments (0827 and 0825). Coverage in the tidal portions of the river proper (0828) was limited to an area on the eastern shore near the mouth.

Table 2-62. Ochlockonee Bay seagrass depth and water clarity targets by segment

Estuary Segment	Depth of Colonization (Z_c) Target (m)	Light Attenuation Coefficient (K_d) Target (1/m)
0825	No target	-
0827	2.8	0.6
0828	No target	-
0829	1.6	1.0
0830	No target	-

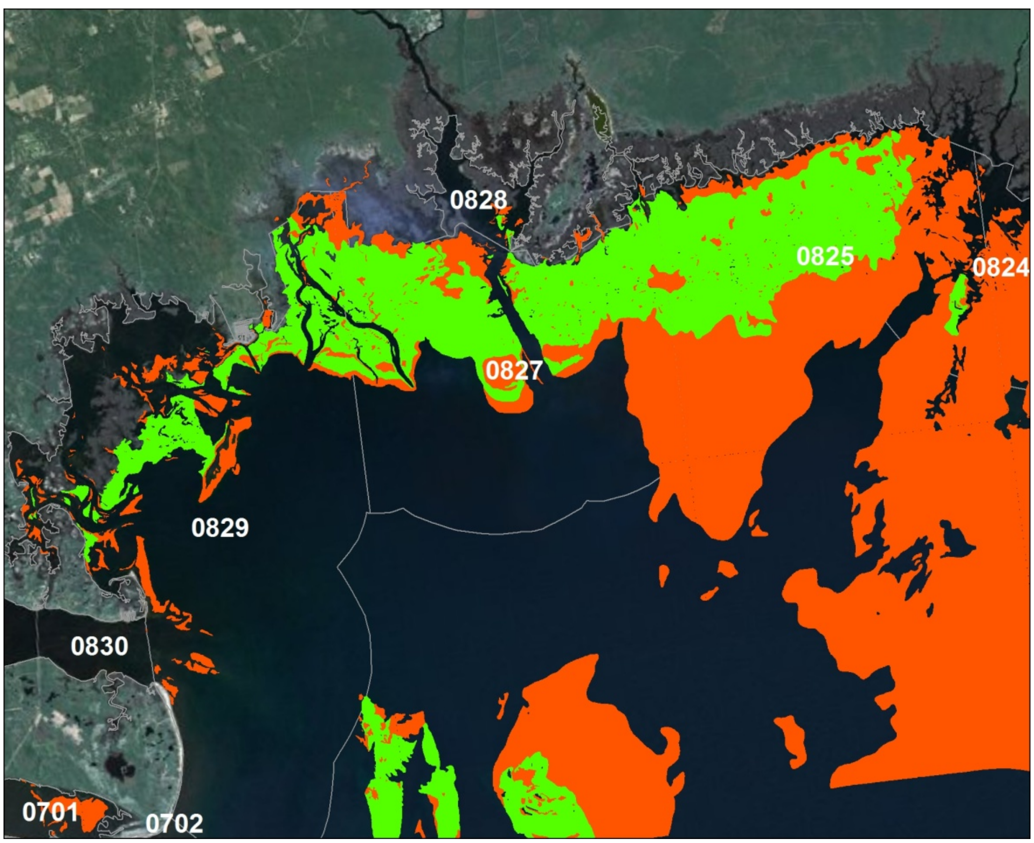

Figure 2-26. Seagrass coverage in the vicinity of Ochlockonee Bay in 1992. Green and orange indicate seagrass delineated as continuous and patchy, respectively.

2.8.5.2. *Chlorophyll* a *Target*

To prevent nuisance algal blooms and protect the estuary's designated uses, chl-a levels must not exceed 20 µg/L more than 10 percent of the time. The rationale for that target is provided in Section 1.2.2.

2.8.5.3. *Dissolved Oxygen Targets*

On the basis of the rationale that sufficient DO is necessary to protect aquatic life, as described in Section 1.2.3, the following DO targets were established:

- Minimum allowable DO of 4.0 mg/L as a water column average for each estuary segment 90 percent of the time over the simulation's time span
- Daily average DO of 5.0 mg/L as a water column average for each estuary segment 90 percent of the time over the simulation's time span
- Minimum 3-hour average DO of 1.5 mg/L in the bottom two layers for each estuary segment over the simulation's time span

2.8.6. Results of Analyses

2.8.6.1. *Mechanistic Model Analysis*

Average load contributions to Ochlockonee Bay from the Apalachee watershed are shown in Table 2-63.

Table 2-63. Average load contributions to Ochlockonee Bay from the Apalachee watershed (2002–2009)

	TN Load[1] (kg/d; Mean ± se)			TP Load[1] (kg/d; Mean ± se)		
Year	Existing[2]	Background[3]	Anthropogenic[4]	Existing[2]	Background[3]	Anthropogenic[4]
2002	4,698 ± 928	4,394 ± 891	304 ± 43	200 ± 31	187 ± 31	13 ± 1
2003	11,111 ± 1,424	10,212 ± 1,325	899 ± 113	406 ± 45	393 ± 45	13 ± 2
2004	5,846 ± 603	5,304 ± 557	542 ± 52	241 ± 22	227 ± 22	15 ± 1
2005	9,102 ± 1,294	8,265 ± 1,197	837 ± 118	344 ± 42	331 ± 42	13 ± 1
2006	2,339 ± 449	2,167 ± 426	172 ± 26	111 ± 19	101 ± 19	11 ± 1
2007	1,552 ± 303	1,374 ± 279	178 ± 29	79 ± 13	70 ± 13	9 ± 1
2008	3,853 ± 1,494	3,466 ± 1,397	387 ± 115	164 ± 48	150 ± 48	13 ± 1
2009	3,435 ± 660	3,142 ± 617	293 ± 50	150 ± 23	136 ± 23	13 ± 1

[1] Annual average daily load from all terminal reaches
[2] Annual average daily load from all terminal reaches computed from existing model scenario
[3] Annual average daily load from all terminal reaches computed from background model scenario (remove anthropogenic sources/land use; use existing hydrology)
[4] Anthropogenic load computed as difference between existing and background load

For Ochlockonee Bay, chl-a targets were met for each segment on the basis of 2002–2009 nutrient loads. DO targets were not met for segment 0828. That segment could not meet the DO targets using either the 2002–2009 nutrient loads scenario or the non-anthropogenic nutrient scenario. The light attenuation coefficient target was not met for either segment 0827 or segment 0829 under either the 2002–2009 nutrient loads or non-anthropogenic nutrient scenario. Table

2-64 identifies which targets were met under 2002–2009 nutrient loads and which targets were not met.

An evaluation of model sensitivity to the biological endpoints applied revealed that the water clarity target was insensitive to changes in nutrients in Ochlockonee Bay.

Table 2-64. Water quality targets met for Ochlockonee Bay based on mechanistic modeling

Segment	DO	Chl-a	K_d
0825	Yes	Yes	No target
0827	Yes	Yes	No
0828	No	Yes	No target
0829	Yes	Yes	No
0830	Yes	Yes	No target

A summary of candidate criteria based on 2002–2009 nutrient loads for Ochlockonee Bay Estuary segments is given in Table 2-65.

Table 2-65. Summary of candidate criteria for Ochlockonee Bay derived from mechanistic modeling

Segment	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
0825	0.79	0.033	2.7
0827	0.51	0.022	1.7
0828	0.55	0.030	1.2
0829	0.47	0.019	1.9
0830	0.66	0.037	1.8

2.8.6.2. Statistical Model Analysis

Data were not sufficient within each segment to conduct statistical analyses in Ochlockonee Bay.

2.8.7. Application of Analyses for Proposed Numeric Nutrient Criteria

EPA evaluated various lines of evidence for deriving numeric nutrient criteria in St. Marks River/St. Marks Offshore. There were insufficient data in St. Marks River/St. Marks Offshore to derive the proposed criteria using statistical models. Therefore, the proposed numeric nutrient criteria were derived using mechanistic modeling output.

EPA evaluated three endpoints in the mechanistic modeling approach for St. Marks River/ St. Marks Offshore: (1) water clarity protective of historic depth of seagrasses, (2) chl-a concentrations associated with balanced phytoplankton biomass, and (3) DO concentrations sufficient to maintain aquatic life. EPA found the water clarity and DO endpoints were not met in all segments under the calibrated 2002–2009 nutrient loads and were insensitive to changes in nutrients loads. As a result, the water clarity and DO endpoints were not used in St. Marks River/St. Marks Offshore. The chl-a target was met and was shown to be sensitive to changes in nutrients. The proposed criteria were derived to be protective of chl-a concentrations. The values under mechanistic modeling represent the 90th percentile annual geometric mean nutrient concentrations from the 2002–2009 modeled nutrient loads.

Proposed numeric nutrient criteria for TN, TP, and chl-a in St. Marks River/St. Marks Offshore segments are summarized in Table 2-66.

Table 2-66. Proposed and candidate numeric nutrient criteria for St. Marks River/St. Marks Offshore segments

		Proposed Criteria			Mechanistic Modeling		
SEGMENT	SEGMENT ID	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
St. Marks River	0828	0.55	0.030	1.2	0.55	0.030	1.2
St. Marks River Offshore	0827	0.51	0.022	1.7	0.51	0.022	1.7

EPA evaluated various lines of evidence for deriving numeric nutrient criteria in Ochlockonee-St. Marks Offshore/Ochlockonee Bay/Ochlockonee Offshore. There were insufficient data in this area to derive the proposed criteria using statistical models. Therefore, the proposed numeric nutrient criteria were derived using mechanistic modeling output.

EPA evaluated three endpoints in the mechanistic modeling approach for Ochlockonee-St. Marks Offshore/Ochlockonee Bay/Ochlockonee Offshore: (1) water clarity protective of historic depth of seagrasses, (2) chl-a concentrations associated with balanced phytoplankton biomass, and (3) DO concentrations sufficient to maintain aquatic life. EPA found that the water clarity target was not met under the calibrated 2002–2009 nutrient loads and was insensitive to changes in nutrients. Therefore, the water clarity endpoint was not used in Ochlockonee-St. Marks Offshore/Ochlockonee Bay/Ochlockonee Offshore. Both the chl-a and DO targets were met under the calibrated 2002–2009 nutrient loads and were shown to be sensitive to changes in nutrients. The proposed criteria were derived to be protective of chl-a and DO concentrations. The values under mechanistic modeling represent the 90th percentile annual geometric mean nutrient concentrations from the 2002–2009 modeled nutrient loads.

Proposed numeric nutrient criteria for TN, TP, and chl-a in Ochlockonee-St. Marks Offshore/Ochlockonee Bay/Ochlockonee Offshore segments are summarized in Table 2-67.

Table 2-67. Proposed and candidate numeric nutrient criteria for Ochlockonee-St. Marks Offshore/Ochlockonee Bay/Ochlockonee Offshore segments

		Proposed Criteria			Mechanistic Modeling		
SEGMENT	SEGMENT ID	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
Ochlockonee Offshore	0829	0.47	0.019	1.9	0.47	0.019	1.9
Ochlockonee Bay	0830	0.66	0.037	1.8	0.66	0.037	1.8
Ochlockonee-St. Marks Offshore	0825	0.79	0.033	2.7	0.79	0.033	2.7

2.8.8. Downstream Protective Values

In Ochlockonee Bay mechanistic models were applied to derive proposed DPVs for TN and TP shown in Table 2-68.

Table 2-68. Proposed DPVs for Ochlockonee Bay

Tributary[a]	LSPC Model Watershed ID	USGS Reach Code	Estuary Segment	TN (mg/L)	TP (mg/L)
West Goose Creek	60052	03120001002608	0829	0.71	0.033
	60053	03120001001724	0829	1.55	0.038
	60054	03120001003384	0829	0.69	0.026
Saint Marks River	60055	03120001000543	0828	0.45	0.041
Wakulla River	60056	03120001000058	0828	0.90	0.050
East River	60057	03120001003016	0828	0.59	0.024
Porpoise Creek	60058	03120001000601	0825	0.76	0.023
Chaires Creek	60155	03120003000397	0830	0.51	0.027
Sopchoppy River	60157	03120003001130	0830	0.83	0.032
Dead River	60158	03120003001179	0830	0.66	0.083

[a] Tributary names left blank are unnamed

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2.9. Big Bend

2.9.1. Proposed Numeric Nutrient Criteria

Proposed numeric nutrient criteria for TN, TP, and chl-a in Big Bend segments are summarized in Table 2-69.

Table 2-69. Proposed numeric nutrient criteria for Big Bend segments

Segment Name	Segment Number	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
Steinhatchee Offshore	0818	0.39	0.032	4.8
Steinhatchee River	0819	0.67	0.077	1.0
Steinhatchee Offshore	0820	0.34	0.018	3.5
Steinhatchee-Fenholloway Offshore	0821	0.40	0.023	4.1
Fenholloway	0822	1.15	0.444	1.9
Fenholloway Offshore	0823	0.48	0.034	10.3
Econfina Offshore	0824	0.59	0.028	4.6
Econfina	0832	0.55	0.032	4.4

2.9.2. General Characteristics

2.9.2.1. System Description

The Big Bend region of Florida is in the northeastern Gulf of Mexico and includes all or part of eight Florida counties (Franklin, Liberty, Gadsden, Leon, Jefferson, Madison, Taylor, and Lafayette) and four Georgia counties (Decatur, Grady, Thomas, and Brooks). Apalachee Bay is part of a broad, shallow shelf area that extends along the entire Gulf Coast of peninsular Florida (Livingston 2010). For the purposes of this system description, the Big Bend region will include two river basins: Aucilla and Econfina/Fenholloway. The Aucilla River Basin and the Econfina/Fenholloway River Basin drain to Apalachee Bay and the Gulf of Mexico (Livingston 2010). Major rivers discharging to the Apalachee Bay and the Gulf of Mexico from the Big Bend region include the Aucilla, Econfina, Fenholloway, and Steinhatchee rivers (FDEP 2010).[50]

The surface area of Apalachee Bay is around 133 mi^2 (344 km^2 [85,112 ac]) (FDEP 2010). Most of the bay is part of the Big Bend Seagrass Aquatic Preserve, which is designated as an OFW.[51] The preserve includes most of the Wakulla and St. Marks rivers southward toward the Steinhatchee River (FDEP 2010). There is limited development in this region of Florida (Livingston et al. 1998). The Econfina River Basin contains two major rivers: the Fenholloway and Econfina. Originating in San Pedro Bay swamp, the Fenholloway River drains approximately 3.3 mi^2 (8.6 km^2 [862 ha]), and the Econfina River (northwest of the Fenholloway River) drains approximately 2.4 mi^2 (6.2 km^2) [619 ha]) of land (Heck 1976; Livingston 2010).

[50] The information presented in this system description was compiled to summarize local information pertaining to the proposed numeric nutrient criteria for this estuary. For more information on EPA's process of delineating, segmenting, and deriving numeric nutrient criteria for this estuary, please see Section 1.3.
[51] Section 62-302.700, F.A.C.

This region usually receives abundant rainfall throughout the year. Annual averages from 1948 to 2008 ranged from a low of approximately 31 in (79 cm) to a high of 104 in (264 cm). Summer (June through August) is generally the wettest season with an average 22 in (56 cm) of rainfall, mainly from tropical storms. The fall months, especially October, typically receive the lowest rainfall during the year (Lewis et al. 2009). In the Big Bend region, El Niño events are often associated with higher than average rainfall and runoff in the winter (Carlson et al. 2010).

The major land uses in the Econfina subwatershed are wetlands (49%), forest (43%), urban (3%), and agricultural (2%) (Fry et al. 2011; SRWMD No date).[52] The Floridan aquifer underlies the Aucilla and Econfina/Fenholloway river basins (USGS 2009).

2.9.2.2. Impaired Waters[53]

Five Class II and Class III marine WBIDs in the Big Bend region are listed for a nutrient-related parameter on Florida's CWA section 303(d) list approved by EPA. Of the five WBIDs, two are Class II WBIDs and three are Class III marine WBIDs. The two Class II WBIDs are impaired for DO (WBIDs 3512 and 3518). The three Class III marine WBIDs are also impaired for DO (WBIDs 3473A, 3573C, and 3705).[54]

One final nutrient-related TMDL for Class II or Class III marine WBIDs exists in the Big Bend area, the *Fenholloway River, Econfina River Basin Nutrient TMDL,* covering Class III marine WBID 3473A.[55]

2.9.2.3. Water Quality

The main sources of nutrients to Apalachee Bay are riverine inputs, which vary as a function of rainfall (FDEP 2010). The results of a 1998–1999 EPA nutrient study of the Econfina and Fenholloway systems showed that DO was lower in the Fenholloway system than the Econfina system. At the Fenholloway River mouth and estuarine zones, average DO levels were 4.4 and 5.6 mg/L, respectively. DO concentrations at the Econfina River mouth and estuarine zone averaged 5.5 and 6.1 mg/L, respectively (USEPA 2000).

Results from three sites offshore of the Aucilla River from the 2001 LakeWatch Program showed mean chl-a concentrations ranged from 0.93 to 1.00 µg/L (FDEP 2010). Chl-a concentrations were higher at the Fenholloway estuarine and nearshore monitoring stations compared to those in

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[52] See Volume 1, Appendix C and its attachments for a summary of how land use data was combined and for a detailed breakdown of land uses.

[53] For more information about the data source, see Appendix A.

[54] The nutrient-related 303(d) list was a compilation of EPA-approved 303(d) listing information for nutrients, chl-a, and DO provided in three decision documents: September 2, 2009, Basin Groups 1, 2, and 5 EPA Decision Document (http://www.epa.gov/region4/water/tmdl/florida/documents/fl09303d_decisiondoc_090209.pdf);
May 13, 2010, Basin Group 3 EPA Decision Document (http://www.epa.gov/region4/water/tmdl/florida/documents/fl303d_%20partialapproval_decision_docs051410.pdf);
and the December 21, 2010, Basin Group 4 EPA Decision Document (http://www.epa.gov/region4/water/tmdl/florida/documents/group_4_final_dec_doc_and_partial_app_letter_12_21_10.pdf).

[55] TMDLs were identified in February 2011 by compiling nutrient-related draft/final TMDLs from the following three sources:
FDEP TMDL website (http://www.dep.state.fl.us/water/tmdl/index.htm)
EPA Region 4 website (http://www.epa.gov/region4/water/tmdl/florida/index.html)
EPA National WATERS expert query tool (http://www.epa.gov/waters/tmdl/expert_query.html).

the Econfina Estuary (Livingston 2010). EPA collected chl-a samples from 69 stations during a 1998–1999 nutrient study of the Econfina and Fenholloway estuaries. Mean chl-a concentrations within the estuaries were 0.7–2.3 µg/L at Econfina sites and 2.0–5.8 µg/L in Fenholloway sites, with a maximum concentration of 10 µg/L in the Fenholloway Estuary (USEPA 2000).

Data from 1998 to 2004 indicated that surface water color was higher in the estuarine and nearshore areas of the Fenholloway River compared to similar areas in the Econfina River. No statistically significant differences in surface water color exist in areas offshore of the two rivers (Livingston 2010). A 1998–1999 EPA study also found water color in the Fenholloway Estuary to be about four times greater than in the Econfina Estuary (130 and 32 PCU, respectively). Average water color found at reference sites in the Fenholloway and Econfina rivers was about 880 PCU and 160 PCU, respectively (USEPA 2000). The median TSS values at Fenholloway River sampling sites ranged from 6 to 8 mg/L between 1983 and 1988. The median TSS values from sampling stations in the Econfina River during the same sampling period ranged from 2 to 11 mg/L (Pescador and Rasmussen 1995). No additional information was found in the available literature.

During the 2001 LakeWatch water quality sampling in Big Bend, mean TN ranged from 0.19 to 0.23 mg/L, and mean TP ranged from 0.0079 to 0.0100 mg/L (FDEP 2010).

Results of quarterly nutrient monitoring from 1991 to 2003 showed that the Fenholloway River was the primary source of ammonia (NH_3) and phosphate to Apalachee Bay because of loading from a pulp mill. From 1992 through 2004, the Fenholloway estuarine stations had significantly higher concentrations of nitrate+nitrite compared to the Econfina estuarine stations (Livingston 2010). In the Aucilla River Basin, the median ammonia-nitrogen concentration was 0.02 mg/L (Ham and Hatzell 1996).

2.9.2.4. Biological Characteristics

The inshore marine regions of the Big Bend region function as highly productive habitats and nurseries for developing stages of offshore organisms. Extensive seagrass beds, a prominent feature of this region, provide important habitat and contribute to primary production. The offshore parts of the bay generally have low nutrient levels, and phytoplankton production is relatively low (Livingston 2010).

Carlson et al. (2010) reviewed aerial seagrass surveys in 1984, 2001, and 2006 to determine changes in seagrass extent in the Big Bend region, including Aucilla, Econfina, Fenholloway, Keaton Beach, Steinhatchee North, Steinhatchee South, Horseshoe West, and Horseshoe East regions. Between 1984 and 2001, total seagrass coverage (including patchy and continuous seagrass) decreased in the Big Bend region by 11 mi^2 (28 km^2 [2,801 ha]). Between 2001 and 2006 total seagrass coverage decreased further by 0.16 mi^2 (0.41 km^2 [41 ha]). Carlson et al. (2010) attributed the seagrass loss in Big Bend to lowered salinities, higher turbidities and color, and increased phytoplankton biomass caused by increased river discharges during storms. Other factors associated with seagrass vulnerability and seagrass loss include elevated nitrogen loads and timing of runoff events (Carlson et al. 2010). Sampling results in 1972–1973 showed that seagrasses in the inshore areas off the Fenholloway River had less macrophyte biomass compared to areas off the Econfina system, largely from the effects of pulp mill effluents (Zimmerman and Livingston 1976).

From 1999 to 2004, the Fenholloway system had higher numbers of phytoplankton cells compared to the Econfina system (with the exception of two offshore stations), and reduced phytoplankton species richness (Livingston 2010). From 1999 to 2004, the Fenholloway system also experienced increased activity by bloom species. In 1999 and 2000, phytoplankton blooms of toxic diatoms and blue-green algae species occurred in the Fenholloway system. In 1999–2000, changes in phytoplankton abundance, species composition, and the occurrence of blooms were influenced by several factors, some of which might have been related to drought conditions. In August 2003 the dominant phytoplankton in the Fenholloway system were the bloom species *Pseudonitzschia pseudodelicatissima* (diatom, potentially toxic species) and *Chattonella* sp. (raphidophyte). High concentrations of those bloom species in the Fenholloway system were attributed to high rainfall and river flow conditions during the summer of 2003 (Livingston 2010). HABs of *K. brevis* have been observed in the Big Bend region and have been associated with fish kills (Carlson and Clarke 2009).

Studies of macroinvertebrates in the 1970s and 1980s have shown that the Econfina Estuary contains greater species diversity and greater numbers of animals than Fenholloway (Dugan and Livingston 1982; Livingston 2010). Another study showed that macrofaunal abundance, diversity, and trophic organization were influenced by macrophyte biomass (Stoner 1980). Lewis (1984) found that the spatial distribution of grass bed crustaceans in Apalachee Bay was influenced by macrophyte biomass. Vegetated microhabitats supported greater abundance and species richness than unvegetated microhabitats (Lewis 1984).

From December 1976 to November 1977 Stoner (1983) examined the relationship between fish assemblages and seagrass biomass in Apalachee Bay off the mouths of the Fenholloway and Econfina rivers. During the 12-month study, 53 species of fish were collected from all four sampling sites. Stoner (1983) found a positive relationship between fish abundance and seagrass standing crop between May and September. Seagrass meadows tended to attract fish by providing shelter and increased availability of food (Stoner 1983). From April 1971 to June 1973, Livingston (1975) examined the effects of pulp mill effluents on fish in the Fenholloway River system by comparing the fish community structure to that of the Econfina River system, which represented an unpolluted drainage system. Results showed a low number of species in the Fenholloway Marsh compared to the Econfina Marsh. Several fish kills were also observed at the mouth of the Fenholloway River during the collection period, in areas that were also affected by pulp mill effluents (Livingston 1975).

For a more detailed summary of this water body, see Volume 1, Appendix A.

2.9.3. Data Used

Several data sources specific to Big Bend were used in addition to those sources described in Section 1.4.3, as summarized in Table 2-70.

Table 2-70. Data sources specific to Big Bend models

Data	Source	Location Used
Springs discharge and water quality data	Florida Department of Environmental Protection (FDEP 2004, 2009; Hornsby and Ceryak 1998; Hornsby and Ceryak 2000)	Apalachicola, Suwannee, Econfina, Waccasassa, Crystal, and Withlacoochee watershed models
FDEP Level III Florida Land Use	Suwannee River Water Management District (SRWMD No date)	Apalachee, Econfina, Suwannee, and Waccasassa watershed models
Climate data	Florida State Climate Center (FSCC 2009) and EarthInfo (EarthInfo 2009)	All watershed models
Municipal and industrial point sources	Georgia Environmental Protection Division (GAEPD No date)	Suwannee and Apalachee watershed models
Water quality data	Georgia Environmental Protection Division (GAEPD 2008)	Suwannee and Apalachee watershed models
Georgia Land Use Trends land use	Natural Resources Spatial Analysis Lab (NARSAL 2008)	Suwannee and Apalachee watershed models
FDEP Level III Florida Land Use	Southwest Florida Water Management District (SWFWMD 2007)	Waccasassa, Crystal, and Withlacoochee watershed models

2.9.4. Segmentation

Big Bend was divided into segments on the basis of the existing WBIDs. In some cases, two or more adjoining WBIDs were combined into one segment when there was homogeneity in seagrass distribution. The segments were limited to a distance of 3 nautical miles from the shoreline, where the estuary numeric nutrient criteria would be applicable. Figure 2-27 shows the resulting segments for Big Bend.

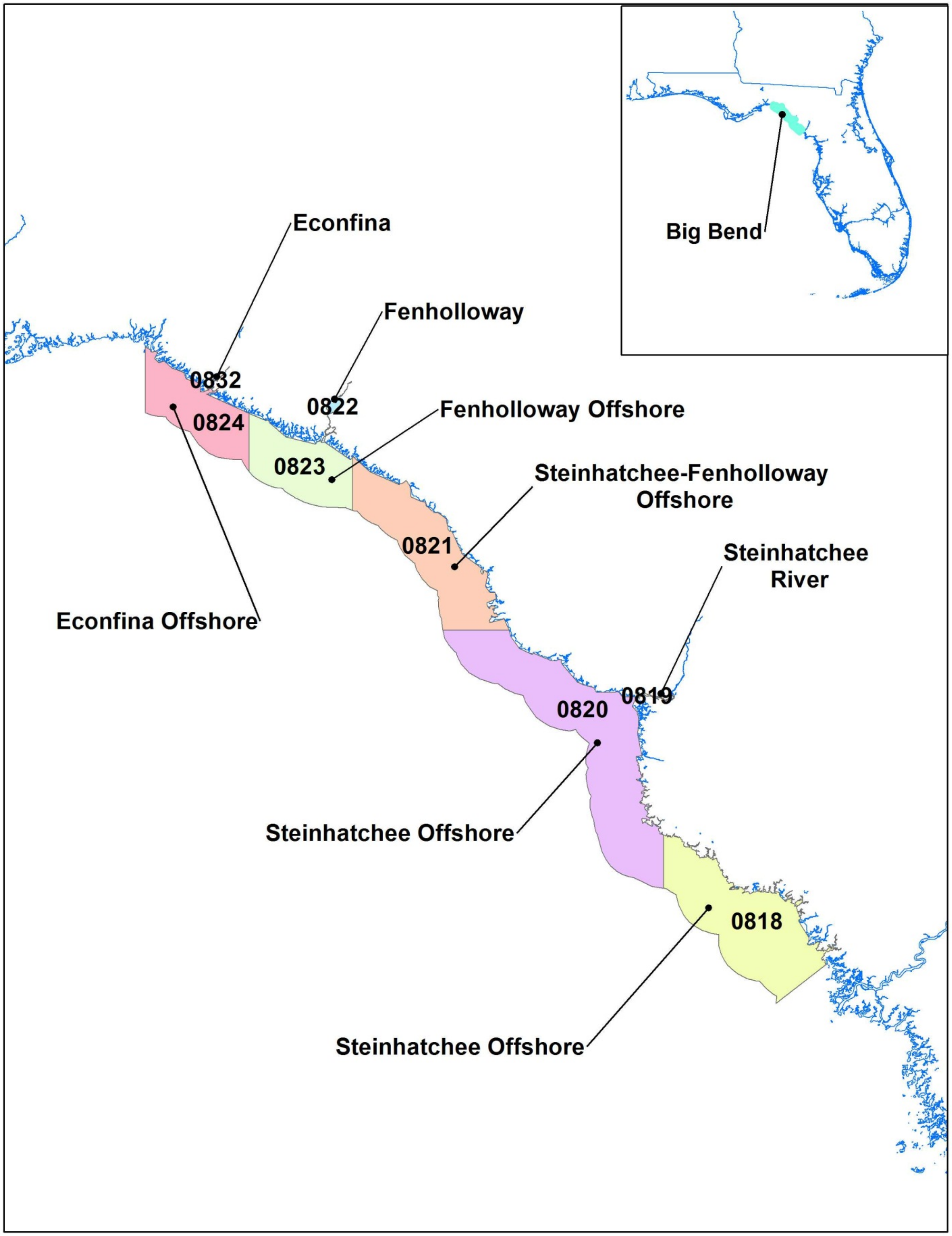

Figure 2-27. Results of Big Bend segmentation

2.9.5. Water Quality Targets

2.9.5.1. Seagrass Depth and Water Clarity Targets

Seagrass depth of colonization (measured as Z_c) and water clarity (measured as K_d) targets were established for the Big Bend Estuary segments (Table 2-71) based on seagrass coverage data from 1992 (Figure 2-28 and Figure 2-29) (FWRI 2012). No seagrass was present within the Fenholloway River (0822), but extensive seagrass was present within the offshore segment, once clear of the plume from the river. Seagrass coverage was present in Apalachee Bay seaward of the 3-nautical-mile limit, but was not considered. There was not adequate CDOM data in this region to evaluate the depth of colonization targets relative to CDOM light attenuation.

Table 2-71. Big Bend seagrass depth and water clarity targets by segment

Estuary Segment	Depth of Colonization (Z_c) Target (m)	Light Attenuation Coefficient (K_d) Target (1/m)
0818	3.5	0.5
0819	No target	-
0820	3.7	0.4
0821	2.6	0.6
0822	No target	-
0823	No target	-
0824	3.1	0.5

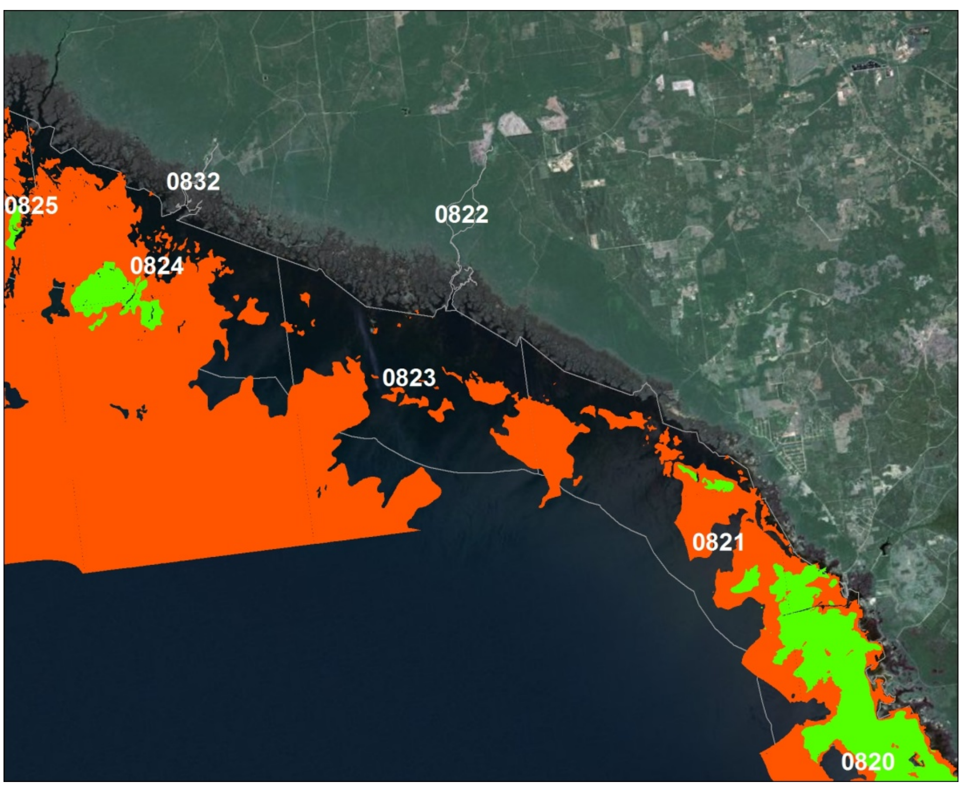

Figure 2-28. Seagrass coverage in 1992 in the vicinity of the Fenholloway River

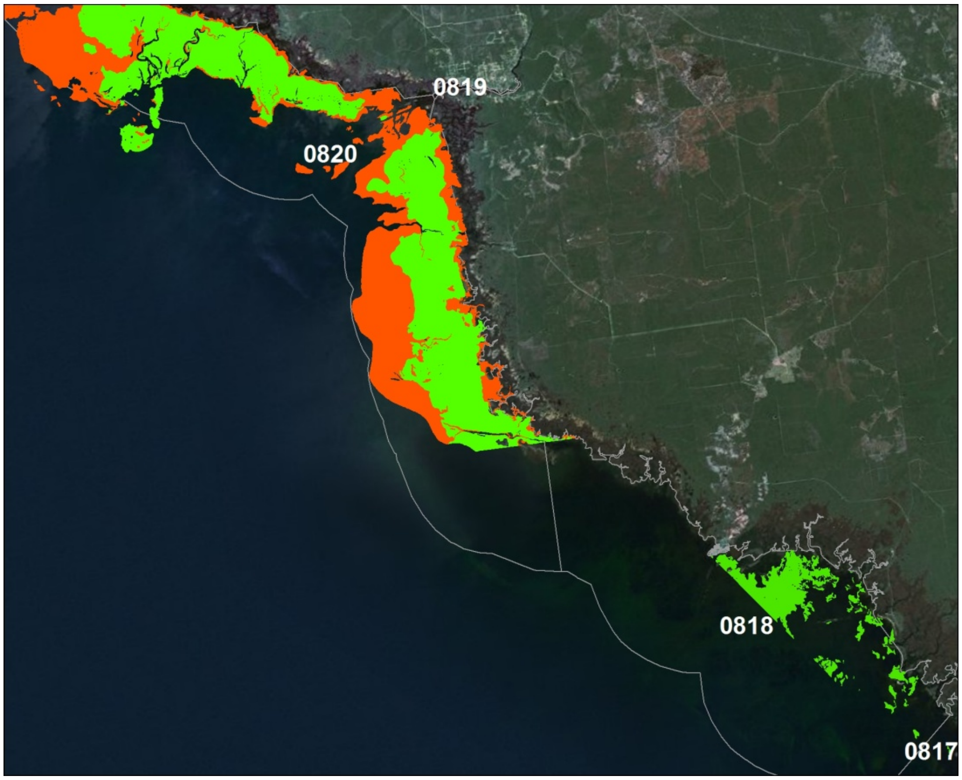

Figure 2-29. Seagrass coverage in 1992 between the Steinhatchee River and the Suwannee River

2.9.5.2. *Chlorophyll* a *Target*

To prevent nuisance algal blooms and protect the estuary's designated uses, chl-a levels must not exceed 20 µg/L more than 10 percent of the time. The rationale for that target is provided in Section 1.2.2.

2.9.5.3. *Dissolved Oxygen Targets*

On the basis of the rationale that sufficient DO is necessary to protect aquatic life, as described in Section 1.2.3, the following DO targets were established:

- Minimum allowable DO of 4.0 mg/L as a water column average for each estuary segment 90 percent of the time over the simulation's time span
- Daily average DO of 5.0 mg/L as a water column average for each estuary segment 90 percent of the time over the simulation's time span
- Minimum 3-hour average DO of 1.5 mg/L in the bottom two layers for each estuary segment over the simulation's time span

2.9.6. Results of Analyses

2.9.6.1. *Mechanistic Model Analysis*

Average load contributions from the Econfina and Apalachee watersheds are shown in Table 2-72 and Table 2-73.

Table 2-72. Average load contributions to Big Bend from the Econfina watershed (2002–2009)

Year	TN Load[1] (kg/d; Mean + se)			TP Load[1] (kg/d; Mean + se)		
	Existing[2]	Background[3]	Anthropogenic[4]	Existing[2]	Background[3]	Anthropogenic[4]
2002	2515 ± 66	1643 ± 59	873 ± 10	453 ± 4	83 ± 3	370 ± 3
2003	6356 ± 291	4941 ± 259	1415 ± 33	615 ± 14	184 ± 10	431 ± 4
2004	5030 ± 262	3872 ± 248	1158 ± 16	564 ± 16	168 ± 15	396 ± 3
2005	4852 ± 169	3636 ± 154	1216 ± 17	560 ± 8	143 ± 6	417 ± 3
2006	2302 ± 82	1471 ± 73	832 ± 11	413 ± 4	74 ± 2	340 ± 2
2007	1546 ± 61	885 ± 55	661 ± 8	366 ± 3	62 ± 2	304 ± 2
2008	2493 ± 308	1685 ± 281	808 ± 28	439 ± 16	91 ± 11	348 ± 6
2009	2739 ± 101	1933 ± 93	806 ± 11	434 ± 5	91 ± 3	343 ± 2

[1] Annual average daily load from all terminal reaches
[2] Annual average daily load from all terminal reaches computed from existing model scenario
[3] Annual average daily load from all terminal reaches computed from background model scenario (remove anthropogenic sources/land use; use existing hydrology)
[4] Anthropogenic load computed as difference between existing and background load

Table 2-73. Average load contributions to Big Bend from the Apalachee watershed (2002–2009)

Year	TN Load[1] (kg/d; Mean + se)			TP Load[1] (kg/d; Mean + se)		
	Existing[2]	Background[3]	Anthropogenic[4]	Existing[2]	Background[3]	Anthropogenic[4]
2002	3165 ± 554	2892 ± 520	273 ± 39	150 ± 22	140 ± 22	10 ± 1
2003	8435 ± 1138	7586 ± 1034	849 ± 111	325 ± 38	316 ± 39	9 ± 2
2004	4811 ± 528	4293 ± 483	518 ± 51	206 ± 20	194 ± 20	12 ± 1
2005	7402 ± 1057	6596 ± 961	806 ± 114	292 ± 36	282 ± 36	10 ± 1
2006	1632 ± 347	1472 ± 324	159 ± 25	85 ± 16	77 ± 16	8 ± 1
2007	1278 ± 268	1106 ± 244	172 ± 29	68 ± 12	60 ± 12	7 ± 0
2008	2682 ± 913	2325 ± 824	357 ± 102	124 ± 32	113 ± 32	11 ± 1
2009	2469 ± 504	2197 ± 463	272 ± 47	116 ± 19	105 ± 19	11 ± 1

[1] Annual average daily load from all terminal reaches
[2] Annual average daily load from all terminal reaches computed from existing model scenario
[3] Annual average daily load from all terminal reaches computed from background model scenario (remove anthropogenic sources/land use; use existing hydrology)
[4] Anthropogenic load computed as difference between existing and background load

For Big Bend, the chl-a target was met for each segment on the basis of 2002–2009 nutrient loads. DO targets were not met for segments 0819 and 0822. Those segments could not meet the DO targets using either the 2002–2009 nutrient loads scenario or the non-anthropogenic nutrient scenario. The light attenuation coefficient target was not met for segments 0818, 0820, 0821, and 0824 under either the 2002–2009 nutrient loads or the non-anthropogenic nutrient scenario. Table 2-74 identifies which targets were met under 2002–2009 nutrient loads and which targets were not met.

An evaluation of model sensitivity to the biological endpoints applied revealed that light and DO targets were insensitive to changes in nutrients in Big Bend.

Table 2-74. Water quality targets met for Big Bend based on mechanistic modeling

Segment	DO	Chl-a	K_d
0818	Yes	Yes	No
0819	No	Yes	No target
0820	Yes	Yes	No
0821	Yes	Yes	No
0822	No	Yes	No target
0823	Yes	Yes	No target
0824	Yes	Yes	No
0832	Yes	Yes	No target

A summary of candidate criteria based on 2002–2009 nutrient loads for Big Bend Estuary segments is given in Table 2-75.

Table 2-75. Summary of candidate criteria for Big Bend derived from mechanistic modeling

Segment	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
0818	0.39	0.032	4.8
0819	0.67	0.077	1.0
0820	0.34	0.018	3.5
0821	0.40	0.023	4.1
0822	1.15	0.444	1.9
0823	0.48	0.034	10.3
0824	0.59	0.028	4.6
0832	0.55	0.032	4.4

2.9.6.2. Statistical Model Analysis

Data were not sufficient within each segment to conduct statistical analyses in Big Bend.

2.9.7. Application of Analyses for Proposed Numeric Nutrient Criteria

EPA evaluated various lines of evidence for deriving numeric nutrient criteria in Steinhatchee Offshore. There were insufficient data in Steinhatchee Offshore to derive the proposed criteria using statistical models. As a result, EPA derived the proposed numeric nutrient criteria for Steinhatchee Offshore using the mechanistic modeling results.

EPA evaluated three endpoints in the mechanistic modeling approach for Steinhatchee Offshore: (1) water clarity protective of historic depth of seagrasses, (2) chl-a concentrations associated with balanced phytoplankton biomass, and (3) DO concentrations sufficient to maintain aquatic life. EPA found that the water clarity target was not met under the calibrated 2002–2009 nutrient loads and was insensitive to changes in nutrients. Therefore, the water clarity endpoint was not used in Steinhatchee Offshore. Both the chl-a and DO targets were met under the calibrated 2002–2009 nutrient loads and were shown to be sensitive to changes in nutrients. The proposed criteria were derived to be protective of chl-a and DO concentrations. The values under mechanistic modeling represent the 90th percentile annual geometric mean nutrient concentrations from the 2002–2009 modeled nutrient loads.

Proposed numeric nutrient criteria for TN, TP, and chl-a in the Steinhatchee Offshore segment are summarized in Table 2-76.

Table 2-76. Proposed and candidate numeric nutrient criteria for the Steinhatchee Offshore segment

		Proposed Criteria			Mechanistic Modeling		
SEGMENT	SEGMENT ID	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
Steinhatchee Offshore	0818	0.39	0.032	4.8	0.39	0.032	4.8

EPA evaluated various lines of evidence for deriving numeric nutrient criteria in Steinhatchee River/Steinhatchee Offshore. There were insufficient data in this area to derive the proposed criteria using statistical models. As a result, EPA derived the proposed numeric nutrient criteria for Steinhatchee River/Steinhatchee Offshore using the mechanistic modeling results.

EPA evaluated three endpoints in the mechanistic modeling approach for Steinhatchee River/ Steinhatchee Offshore: (1) water clarity protective of historic depth of seagrasses, (2) chl-a concentrations associated with balanced phytoplankton biomass, and (3) DO concentrations sufficient to maintain aquatic life. EPA found the water clarity and DO endpoints were not met in all segments under the calibrated 2002–2009 nutrient loads and were insensitive to changes in nutrients loads. As a result, the water clarity and DO endpoints were not used in Steinhatchee River/Steinhatchee Offshore. The chl-a target was met and was shown to be sensitive to changes in nutrients. The proposed criteria were derived to be protective of chl-a concentrations. The values under mechanistic modeling represent the 90th percentile annual geometric mean nutrient concentrations from the 2002–2009 modeled nutrient loads.

Proposed numeric nutrient criteria for TN, TP, and chl-a in Steinhatchee River/Steinhatchee Offshore segments are summarized in Table 2-77.

Table 2-77. Proposed and candidate numeric nutrient criteria for Steinhatchee River/Steinhatchee Offshore segments

		Proposed Criteria			Mechanistic Modeling		
SEGMENT	SEGMENT ID	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
Steinhatchee River	0819	0.67	0.077	1.0	0.67	0.077	1.0
Steinhatchee Offshore	0820	0.34	0.018	3.5	0.34	0.018	3.5

EPA evaluated various lines of evidence for deriving numeric nutrient criteria in Steinhatchee-Fenholloway Offshore. There were insufficient data in Steinhatchee-Fenholloway Offshore to derive the proposed criteria using statistical models. As a result, EPA derived the proposed numeric nutrient criteria for Steinhatchee-Fenholloway Offshore using the mechanistic modeling results.

EPA evaluated three endpoints in the mechanistic modeling approach for Steinhatchee-Fenholloway Offshore: (1) water clarity protective of historic depth of seagrasses, (2) chl-a concentrations associated with balanced phytoplankton biomass, and (3) DO concentrations sufficient to maintain aquatic life. EPA found that the water clarity target was not met under the calibrated 2002–2009 nutrient loads and was insensitive to changes in nutrients. Therefore, the water clarity endpoint was not used in Steinhatchee Offshore. Both the chl-a and DO targets were met under the calibrated 2002–2009 nutrient loads and were shown to be sensitive to changes in nutrients. The proposed criteria were derived to be protective of chl-a and DO concentrations. The values under mechanistic modeling represent the 90th percentile annual geometric mean nutrient concentrations from the 2002–2009 modeled nutrient loads.

Proposed numeric nutrient criteria for TN, TP, and chl-a in the Steinhatchee-Fenholloway Offshore segment are summarized in Table 2-78.

Table 2-78. Proposed and candidate numeric nutrient criteria for the Steinhatchee-Fenholloway Offshore segment

		Proposed Criteria			Mechanistic Modeling		
SEGMENT	SEGMENT ID	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
Steinhatchee-Fenholloway Offshore	0821	0.40	0.023	4.1	0.40	0.023	4.1

EPA evaluated various lines of evidence for deriving numeric nutrient criteria in Fenholloway/ Fenholloway Offshore. There were insufficient data in Fenholloway/Fenholloway Offshore to derive the proposed criteria using statistical models. As a result, EPA derived the proposed numeric nutrient criteria for Fenholloway/Fenholloway Offshore using the mechanistic modeling results.

Because depth of colonization targets were not available in Fenholloway/Fenholloway Offshore, EPA evaluated two endpoints in the mechanistic modeling approach: (1) chl-a concentrations associated with balanced phytoplankton biomass and (2) DO concentrations sufficient to maintain aquatic life. EPA found that the DO target was not met under the calibrated 2002–2009 nutrient loads and was insensitive to changes in nutrients. Therefore, the DO endpoint was not used in Fenholloway/Fenholloway Offshore. However, the chl-a target was met under the calibrated 2002–2009 nutrient loads and was sensitive to changes in nutrients. The proposed criteria were derived to be protective of chl-a concentrations. The values under mechanistic modeling represent the 90th percentile annual geometric mean nutrient concentrations from the 2002–2009 modeled nutrient loads.

Proposed numeric nutrient criteria for TN, TP, and chl-a in Fenholloway/Fenholloway Offshore segments are summarized in Table 2-79.

Table 2-79. Proposed and candidate numeric nutrient criteria for Fenholloway/Fenholloway Offshore segments

		Proposed Criteria			Mechanistic Modeling		
SEGMENT	SEGMENT ID	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
Fenholloway	0822	1.15	0.444	1.9	1.15	0.444	1.9
Fenholloway Offshore	0823	0.48	0.034	10.3	0.48	0.034	10.3

EPA evaluated various lines of evidence for deriving numeric nutrient criteria in Econfina/ Econfina Offshore. There were insufficient data in this area to derive the proposed criteria using statistical models. As a result, EPA derived the proposed numeric nutrient criteria for Econfina/Econfina Offshore using the mechanistic modeling results.

EPA evaluated three endpoints in the mechanistic modeling approach for Econfina/Econfina Offshore: (1) water clarity protective of historic depth of seagrasses, (2) chl-a concentrations associated with balanced phytoplankton biomass, and (3) DO concentrations sufficient to maintain aquatic life. EPA found that the water clarity target was not met under the calibrated 2002–2009 nutrient loads and was insensitive to changes in nutrients. Therefore, the water clarity endpoint was not used in Econfina/Econfina Offshore. Both the chl-a and DO targets were met under the calibrated 2002–2009 nutrient loads and were shown to be sensitive to changes in nutrients. The proposed criteria were derived to be protective of chl-a and DO concentrations. The values under mechanistic modeling represent the 90th percentile annual geometric mean nutrient concentrations from the 2002–2009 modeled nutrient loads.

Proposed numeric nutrient criteria for TN, TP, and chl-a in Econfina/Econfina Offshore segments are summarized in Table 2-80.

Table 2-80. Proposed and candidate numeric nutrient criteria for Econfina/Econfina Offshore segments

		Proposed Criteria			Mechanistic Modeling		
SEGMENT	SEGMENT ID	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
Econfina Offshore	0824	0.59	0.028	4.6	0.59	0.028	4.6
Econfina	0832	0.55	0.032	4.4	0.55	0.032	4.4

2.9.8. Downstream Protective Values

In Big Bend mechanistic models were applied to derive proposed DPVs for TN and TP shown in Table 2-81.

Table 2-81. Proposed DPVs for Big Bend

Tributary[a]	LSPC Model Watershed ID	USGS Reach Code	Estuary Segment	TN (mg/L)	TP (mg/L)
Aucilla River	60024	03110103000337	0824	0.75	0.048
Pinhook River	60201	03120001003358	0825	0.72	0.025
Sand Creek	100001	03110102002296	0821	0.72	0.038
Regular Creek	100002	03110102014966	0823	0.95	0.013
Econfina River	100003	03110102000289	0832	0.55	0.039
	100025	03110102000559	0824	0.96	0.011
Spring Warrior Creek	100026	03110102000201	0821	0.80	0.043
Fenholloway River	100027	03110102014951	0822	2.06	0.868
Blue Creek	100028	03110102022239	0821	0.58	0.048
Clearwater Creek	100029	03110102022245	0821	0.32	0.051
	100030	03110102015253	0820	0.39	0.014
Salt Creek	100031	03110102022211	0820	0.83	0.011
Steinhatchee River	100032	03110102012357	0819	0.77	0.045
Rocky Creek	100033	03110102000053	0820	0.39	0.015
Little Rocky Creek	100034	03110102012670	0820	0.43	0.015
Amason Creek	100035	03110102001168	0818	0.46	0.019
Johnson Creek	100036	03110102001183	0818	0.79	0.026

[a] Tributary names left blank are unnamed

2.9.9. References

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2.10. Suwannee Sound

2.10.1. Proposed Numeric Nutrient Criteria

Proposed numeric nutrient criteria for TN, TP, and chl-a in the Suwannee Sound segment are summarized in Table 2-82.

Table 2-82. Proposed numeric nutrient criteria for Suwannee Sound segments

Segment Name	Segment Number	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
Suwannee Offshore	0817	0.78	0.049	5.2

2.10.2. General Characteristics

2.10.2.1. System Description

The Suwannee Estuary is an open estuary with an average depth of 7.2 ft (2.2 m) (FDEP 2010). The Suwannee watershed contains the Suwannee River, which originates in Georgia's Okefenokee Swamp, and drains around 11,043 mi^2 (28,600 km^2) (FDEP 2010; USGS 2004). Within the estuary are Suwannee Sound, which is considered the main estuarine zone, and Horseshoe Cove (Edwards and Raabe 2004; FDEP 2010). [56] FDEP has designated several water bodies in the Suwannee watershed as OFWs. Waters in the Suwannee watershed designated as OFWs include Suwannee River, Lower Suwannee National Wildlife Refuge, Okefenokee National Wildlife Refuge, Suwannee River State Park, Santa Fe River system (portions), and the Withlacoochee River system (portions).[57]

The Suwannee watershed has two major physiographic regions, the Northern Highlands and the Gulf Coastal Lowlands. Most of the lowland area is a karst plain topography containing sinkholes and natural limestone springs. The coastal area has mostly marshes and swamps (Hallas and Magley 2008). The upper areas of the watershed have numerous streams, lakes, and wetlands. The upper two-thirds of the Suwannee River Basin has soils with clay and fine sediments that are resistant to surface water infiltration. The lower third of the basin has a thin layer of highly porous sands that lies over the Floridan aquifer system, and has very few surface waters (USGS 2004).

The Suwannee watershed contains numerous rivers, streams, springs, cypress ponds, swamps, and estuaries. The Suwannee River contributes the majority of freshwater input to Suwannee Sound (FDEP 2010). The upper portion of the river is an acidic, blackwater stream. The river becomes increasingly clear and alkaline downstream as it receives water from the Floridan aquifer via springs and seeps (Hallas and Magley 2008). Major tributaries to the Suwannee River include the Santa Fe, Alapaha, and Withlacoochee rivers (FDEP 2003). The Suwannee River has three major discharge points to the Gulf of Mexico (Brooks and Sulak 2004).

[56] The information presented in this system description was compiled to summarize local information pertaining to the proposed numeric nutrient criteria for this estuary. For more information on EPA's process of delineating, segmenting, and deriving numeric nutrient criteria for this estuary, please see Section 1.3.
[57] Section 62-302.700, F.A.C.

The Suwannee Estuary is dominated by river flow, rather than by ocean water, and its biological communities are highly influenced by freshwater inputs (FDEP 2010). The combination of river flow, groundwater flow, and tidal freshwater advection make the system complex both physically and ecologically. The dynamics also change with the season, adding even more complexity (Edwards and Raabe 2004). According to FDEP, the Suwannee River watershed has the highest density of springs in the world (FDEP 2003). Ninety-eight known springs discharge to the Suwannee River coastal drainage area. Springs greatly influence the flow of the rivers during periods of low rainfall; wetlands influence the flow during periods of high rainfall (FDEP 2010).

Growth and development is limited along the watershed's rivers because of floodplain management ordinances, land use plans, and land acquisition programs. Forested areas cover a large portion of the watershed, with silviculture and agriculture being dominant land uses. Phosphate mining has changed much of the original landscape in southeastern Hamilton County (FDEP 2010). Approximately 45 percent of the watershed is forested or clear-cut, 25 percent is agriculture, 22 percent is wetlands, and 7 percent is urban (Fry et al. 2011; NARSAL 2008; SJRWMD 2006; SRWMD No date).[58]

2.10.2.2. Impaired Waters[59]

One Class III marine WBID in Suwannee Sound is listed for a nutrient-related parameter on Florida's CWA section 303(d) list approved by EPA. That WBID is impaired for nutrients and chl-a (WBID 3422D). No Class II WBIDs with nutrient-related impairments are documented for this area.[60]

No Class II or Class III marine WBIDs with nutrient-related TMDLs were documented for this region.[61]

2.10.2.3. Water Quality

Numerous studies and monitoring efforts have taken place in the Suwannee Estuary. Those include efforts by the University of Florida (Project COAST), the Suwannee River Water Management District (SRWMD), and Florida's Inshore Marine Monitoring and Assessment Program (FDEP 2010). Sample results from the monitoring studies are briefly summarized below.

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[58] See Volume 1, Appendix C and its attachments for a summary of how land use data was combined and for a detailed breakdown of land uses.

[59] For more information about the data source, see Volume 1, Appendix A.

[60] The nutrient-related 303(d) list was a compilation of EPA-approved 303(d) listing information for nutrients, chl-a, and DO provided in three decision documents: September 2, 2009, Basin Groups 1, 2, and 5 EPA Decision Document (http://www.epa.gov/region4/water/tmdl/florida/documents/fl09303d_decisiondoc_090209.pdf);
May 13, 2010, Basin Group 3 EPA Decision Document (http://www.epa.gov/region4/water/tmdl/florida/documents/fl303d_%20partialapproval_decision_docs051410.pdf);
and the December 21, 2010, Basin Group 4 EPA Decision Document (http://www.epa.gov/region4/water/tmdl/florida/documents/group_4_final_dec_doc_and_partial_app_letter_12_21_10.pdf).

[61] TMDLs were identified in February 2011 by compiling nutrient-related draft/final TMDLs from the following three sources:
FDEP TMDL website (http://www.dep.state.fl.us/water/tmdl/index.htm)
EPA Region 4 website (http://www.epa.gov/region4/water/tmdl/florida/index.html)
EPA National WATERS expert query tool (http://www.epa.gov/waters/tmdl/expert_query.html).

Between 2000 and 2007, DO levels from samples in the Suwannee and Santa Fe rivers ranged from 0.9 to 11.9 mg/L. Most high values were observed in the spring and summer (Hallas and Magley 2008). For springs that are in the Suwannee River watershed, the median DO ranged from 0.3 to 3.5 mg/L between 2001 and 2006 (Harrington et al. 2010).

In the Suwannee Estuary, the annual geometric mean chl-a (uncorrected) from 1997 to 2009 was around 8.5 µg/L, as calculated for Project COAST, a monthly monitoring program. During a similar time frame (1995–2009), SRWMD found that the annual geometric mean for chl-a in the Suwannee Estuary ranged from non-detect to 6 µg/L, with no distinct trend found. In 2004 and 2005 chl-a levels in the Suwannee Estuary (in WBID 3422D) were more than 50 percent higher than the historic (1997, 1998, 2002, 2004-2006) minimum concentration of 4.0 µg/L (FDEP 2010).

Turbidity data were not found in the available literature for the Suwannee Estuary. However, some were found for the springs in the Suwannee River watershed. Harrington et al. (2010) reported that median turbidity ranged from 0.1 to 0.375 mg/L in those springs between 2001 and 2006.

In the Suwannee Estuary, the geometric mean TN from 1997 to 2009 was approximately 0.5 mg/L, but it ranged from 0.4 to 0.8 mg/L in nearshore sites and 0.25 to 0.4 mg/L in offshore sites, as analyzed by FDEP (FDEP 2010). Data from SRWMD show the annual geometric mean for TN in the Suwannee Estuary ranged from 0.6 to 1.5 mg/L from 1996 to 2009 (FDEP 2010).

Whereas nitrate (NO_3) concentrations vary throughout the watershed, some springs in the Suwannee watershed (Fanning, Troy, Lafayette Blue, Manatee, and Devils Ear) have some of the highest nitrate concentrations in Florida. Nitrate in the basin is linked to agricultural areas that overlay groundwater resources with karst geology (FDEP 2010).

According to Hallas and Magley (2008), TP in the Suwannee watershed peaked in 1983, with concentrations between 0.3 mg/L (Lower Santa Fe River) and 0.42 mg/L (Middle Suwannee River), calculated as a 3-year rolling average. Monitoring up until 2007 showed that TP concentrations have been steadily declining since that 1983 peak. The trend has not been linked to mining activity, climate, or river flow (Hallas and Magley 2008). In the Suwannee Estuary, the geometric mean TP from Project COAST monitoring data from 1997 to 2007 ranged from around 0.04 to 0.08 mg/L in nearshore sites and 0.02 to 0.04 mg/L in offshore sites (FDEP 2010). During a similar time frame (1995–2009), SRWMD found that the annual geometric mean for TP in the Suwannee Estuary ranged from 0.04 to 0.13 mg/L (FDEP 2010).

Springs in the Suwannee River Basin have high phosphate concentrations (compared to springs in other areas of Florida), attributed to leaching from the Hawthorne Foundation, which is a high-phosphate, marine clay complex (FDEP 2010). The median phosphate concentration for the springs between 2001 and 2006 ranged from 0.022 to 0.085 mg/L (Harrington et al. 2010).

2.10.2.4. Biological Characteristics

The upper two-thirds of the Suwannee River Basin has numerous wetlands, but the lower basin areas closer to the Gulf of Mexico have very few (15% of land use is wetlands) (FDEP 2010;

USGS 2004). The wetland areas in the upper watershed impact river flow during high rainfall events (FDEP 2010).

Phytoplankton production in Suwannee Sound is relatively high (Quinlan et al. 2009). A gradient of phytoplankton from the river to the open estuary is apparent (Carlson et al. 2010; FDEP 2010). A red tide event that occurred from September 2005 to January 2006 was the largest recorded in the Suwannee Estuary, although its cause is unknown (FDEP 2010; Smith 2009).

High macroalgae biomass was observed during the winter and spring of 1999–2001 near the mouth of the Suwannee River; and was attributed to reduced flow, seasonal changes in water clarity, and tidal cycles (FDEP 2010). Blooms remain in the estuary and often near the mouth of the Suwannee River during drier weather and move offshore during the rainy season. Large algal mats are also found in the Suwannee River (FDEP 2003).

The largest population of gulf sturgeon (*Acipenser oxyrinchus desotoi*) is found in the Suwannee River (Sulak and Clugston 1999).

For a more detailed summary of this water body, see Volume 1, Appendix A.

2.10.3. Data Used

Several data sources specific to Suwannee Sound were used in addition to those sources described in Section 1.4.3, as summarized in Table 2-83.

Table 2-83. Data sources specific to Suwannee Sound models

Data	Source	Location Used
Springs discharge and water quality data	Florida Department of Environmental Protection (FDEP 2004, 2009; Hornsby and Ceryak 1998)	Apalachicola, Suwannee, Econfina, Waccasassa, Crystal, and Withlacoochee watershed models
FDEP Level III Florida Land Use	Suwannee River Water Management District (SRWMD No date)	Apalachee, Econfina, Suwannee, and Waccasassa watershed models
Climate data	Florida State Climate Center (FSCC 2009) and EarthInfo (EarthInfo 2009)	All watershed models
Municipal and industrial point sources	Georgia Environmental Protection Division (GAEPD No date)	Suwannee and Apalachee watershed models
Water quality data	Georgia Environmental Protection Division (GAEPD 2008)	Suwannee and Apalachee watershed models
Georgia Land Use Trends land use	Natural Resources Spatial Analysis Lab (NARSAL 2008)	Suwannee and Apalachee watershed models
FDEP Level III Florida Land Use	Southwest Florida Water Management District (SWFWMD 2007)	Waccasassa, Crystal, and Withlacoochee watershed models

2.10.4. Segmentation

An existing WBID (3422D) was used as a segment for Suwannee Sound. The segment was limited to a distance of 3 nautical miles from the shoreline, where the estuary numeric nutrient criteria would be applicable. Figure 2-30 shows the resulting segment for Suwannee Sound.

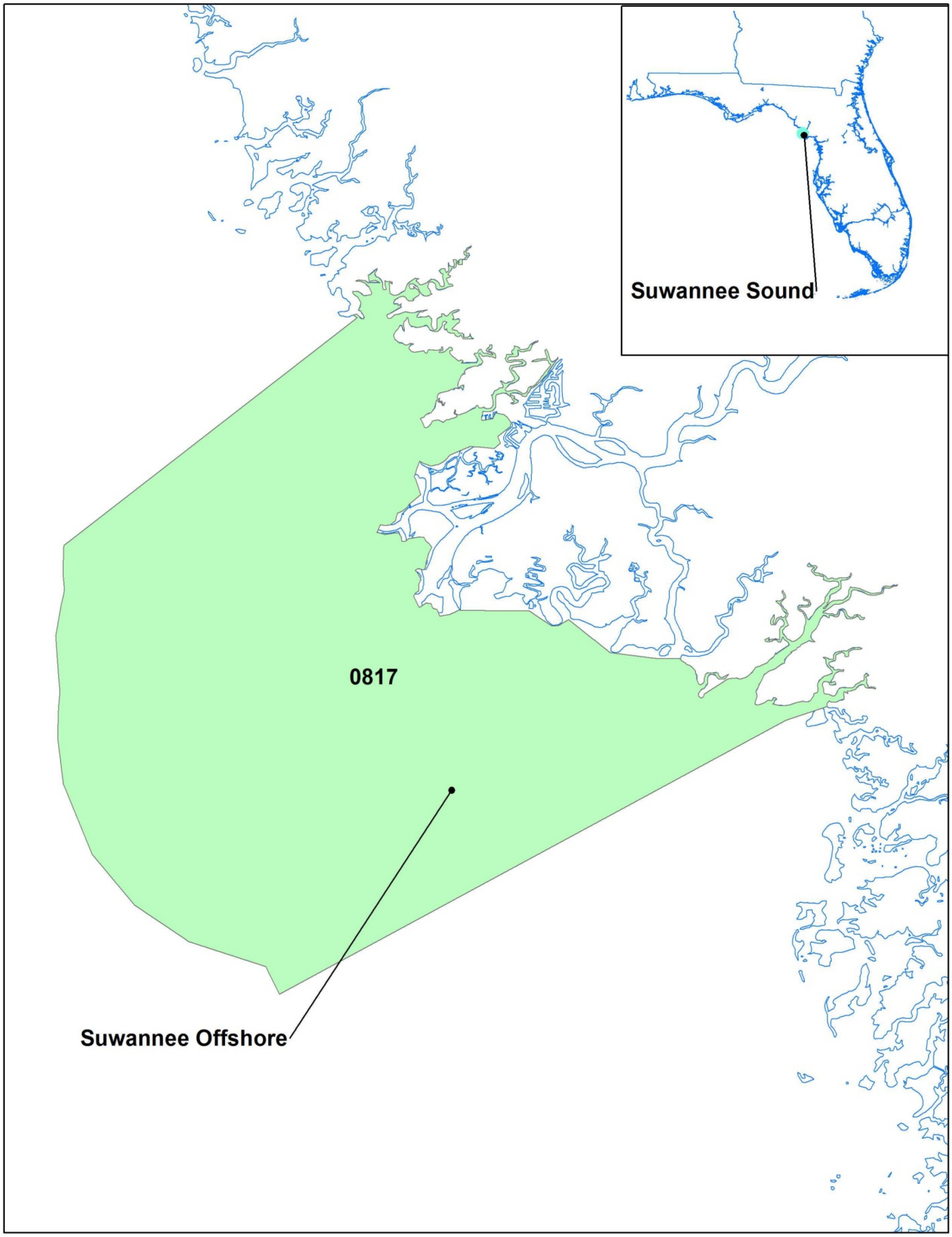

Figure 2-30. Results of Suwannee Sound segmentation

2.10.5. Water Quality Targets

2.10.5.1. Seagrass Depth and Water Clarity Targets

A seagrass depth of colonization (measured as Z_c) target and a water clarity (measured as K_d) target were established for the Suwannee Sound Estuary segment, as shown in Table 2-84 based on a 2001 seagrass coverage (Figure 2-31) (FWRI 2012).

Table 2-84. Suwannee Sound seagrass depth and water clarity targets by segment

Estuary Segment	Depth of Colonization (Z_c) Target (m)	Light Attenuation Coefficient (K_d) Target (1/m)
0817	1.0	1.6

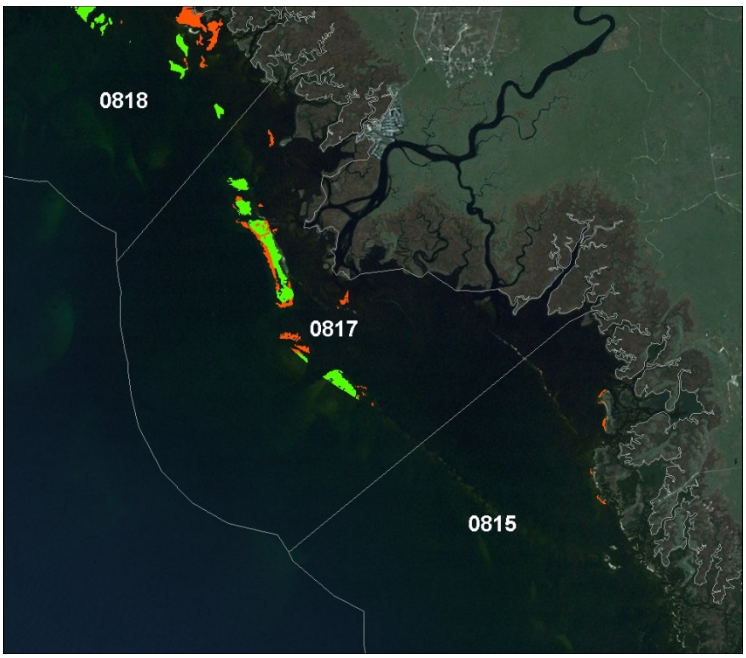

Figure 2-31. Seagrass coverage in the vicinity of the Suwannee River in 2001. Green and orange indicate continuous and patchy seagrass coverage, respectively.

2.10.5.2. Chlorophyll a Target

To prevent nuisance algal blooms and protect the estuary's designated uses, chl-a levels must not exceed 20 µg/L more than 10 percent of the time. The rationale for that target is provided in Section 1.2.2.

2.10.5.3. Dissolved Oxygen Targets

On the basis of the rationale that sufficient DO is necessary to protect aquatic life, as described in Section 1.2.3, the following DO targets were established:

- Minimum allowable DO of 4.0 mg/L as a water column average for each estuary segment 90 percent of the time over the simulation's time span

- Daily average DO of 5.0 mg/L as a water column average for each estuary segment 90 percent of the time over the simulation's time span
- Minimum 3-hour average DO of 1.5 mg/L in the bottom two layers for each estuary segment over the simulation's time span

2.10.6. Results of Analyses

2.10.6.1. Mechanistic Model Analysis

Average load contributions for Suwannee Sound/Cedar Keys watershed are shown in Table 2-85.

Table 2-85. Average load contributions from the Springs Coast watershed

	TN Load[1] (kg/d; Mean + se)			TP Load[1] (kg/d; Mean + se)		
Year	Existing[2]	Background[3]	Anthropogenic[4]	Existing[2]	Background[3]	Anthropogenic[4]
2002	12427 ± 689	8156 ± 408	4271 ± 287	2267 ± 46	542 ± 17	1725 ± 33
2003	24695 ± 1190	14757 ± 627	9938 ± 570	2957 ± 90	834 ± 28	2124 ± 72
2004	25129 ± 1936	15927 ± 1169	9203 ± 769	3561 ± 208	887 ± 60	2674 ± 160
2005	25045 ± 1034	15422 ± 612	9624 ± 428	3962 ± 110	858 ± 40	3104 ± 86
2006	13095 ± 593	8466 ± 321	4628 ± 277	2003 ± 71	518 ± 12	1485 ± 64
2007	9581 ± 277	6571 ± 160	3010 ± 120	1400 ± 24	473 ± 8	927 ± 18
2008	18434 ± 1034	11567 ± 604	6867 ± 439	2058 ± 61	636 ± 25	1423 ± 38
2009	15463 ± 809	9669 ± 437	5793 ± 375	2330 ± 77	615 ± 22	1715 ± 60

[1] Annual average daily load from all terminal reaches
[2] Annual average daily load from all terminal reaches computed from existing model scenario
[3] Annual average daily load from all terminal reaches computed from background model scenario (remove anthropogenic sources/land use; use existing hydrology)
[4] Anthropogenic load computed as difference between existing and background load

In the Suwannee Sound estuary, DO, chl-a, and light attenuation coefficient targets were met using 2002–2009 nutrient loads. Table 2-86 identifies which targets were met under 2002–2009 nutrient loads and which targets were not met.

An evaluation of model sensitivity to the biological endpoints applied revealed that all targets were sensitive to changes in nutrients in Suwannee Sound Estuary.

Table 2-86. Water quality targets met for Suwannee Sound based on mechanistic modeling

Segment	DO	Chl-a	K_d
0817	Yes	Yes	Yes

A summary of candidate criteria for Suwannee Sound Estuary is given in Table 2-87. Nutrient loads from 2002 to 2009 were used to calculate candidate criteria for Suwannee Sound.

Table 2-87. Summary of candidate criteria for Suwannee Sound derived from mechanistic modeling

Segment	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
0817	0.45	0.052	4.2

2.10.6.2. Statistical Model Analysis

Analysis of available data indicated that water clarity in Suwannee Sound was not sensitive to changes in chl-a and nutrient concentrations, likely because of the influence of suspended sediments and CDOM from the Suwannee River. The frequency of algal blooms did increase with increased chl-a, but the candidate criterion associated with a bloom frequency of 10 percent was higher than the upper bound of the range of observed data (Figure 2-32). Hence, a candidate chl-a criteria of 5.2 µg/L was set at the upper bound of the observed data.

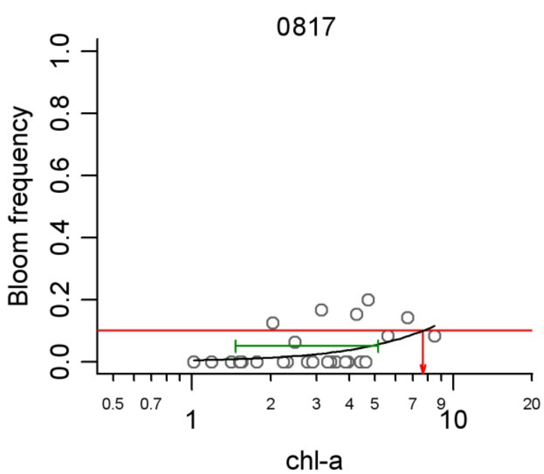

Figure 2-32. Modeled relationship between bloom frequency and TN, TP, and chl-a in Suwannee Sound. Solid black line: modeled mean relationship. Open circles: observed annual geometric means. Dashed horizontal line: 10 percent bloom frequency endpoint. Green line segment: 5th to 95th percentile range of observed data.

Chl-a concentrations exhibited an increasing relationship with increased concentrations of TN and TP (Figure 2-33). Candidate TN and TP criteria derived such that mean chl-a concentrations within the segment were the same as the candidate chl-a criterion value were both less than the lower bound of observed TN and TP concentrations. Hence, candidate TN (0.78 mg/L) and TP (0.049 mg/L) criteria were set at the lower bound of observed annual geometric mean concentrations.

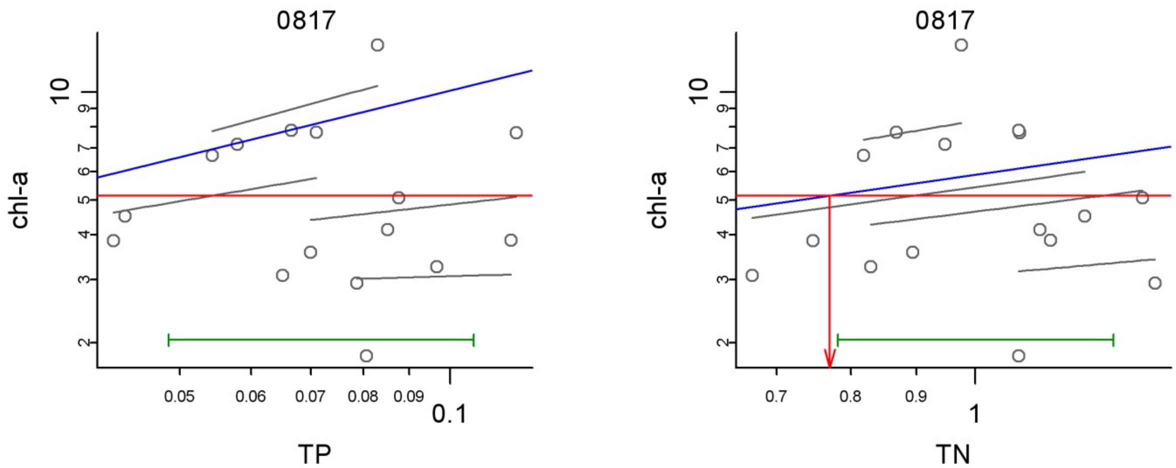

Figure 2-33. Relationships between TN, TP, and chl-a in Suwannee Sound. Open circles: observed annual average values of TN, TP, and chl-a, red horizontal line: chl-a criterion, red vertical arrow: TN and TP criteria associated with chl-a criterion, blue line: estimated segment-wide relationship between TN, TP, and chl-a, grey lines: estimated station-specific relationships between TN, TP, and chl-a, green line segment: 5th to 95th percentile range of observed annual geometric mean TN and TP concentrations.

2.10.7. Application of Analyses for Proposed Numeric Nutrient Criteria

EPA evaluated various lines of evidence for deriving numeric nutrient criteria in Suwannee Sound. Data were sufficient to use statistical analyses as the primary line of evidence when deriving criteria.

Statistically, EPA found that water clarity was not sensitive to changes in nutrient concentration, so criteria were based on the estimated relationship between bloom frequency and nutrient concentrations. Because the candidate criteria derived from bloom frequency were lower than the lower bound of the observed range of data, proposed criteria are based on the lower bound of observed data.

EPA evaluated three endpoints in the mechanical modeling approach for Suwannee Sound: (1) water clarity protective of historic depth of seagrasses, (2) chl-a concentrations associated with balanced phytoplankton biomass, and (3) DO concentrations sufficient to maintain aquatic life. EPA found that all three targets were achieved under the calibrated 2002–2009 nutrient loads and were sensitive to changes in nutrients. The proposed criteria were derived to be protective of all three. The values under mechanistic modeling represent the 90th percentile annual geometric mean nutrient concentrations from the 2002–2009 modeled nutrient loads.

In comparing the mechanistic and statistical model results, EPA found the results to be comparable and corroborative. Because there was sufficient empirical information, EPA relied upon the statistical model results, which are a direct measure of the water quality conditions in the estuary, to derive the proposed criteria for Suwannee Sound.

Proposed numeric nutrient criteria for TN, TP, and chl-a in the Suwannee Sound segment are summarized in Table 2-88.

Table 2-88. Proposed and candidate numeric nutrient criteria for the Suwannee Sound segment

		Proposed Criteria			Mechanistic Modeling			Statistical Modeling		
SEGMENT	SEGMENT ID	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
Suwannee Offshore	0817	0.78	0.049	5.2	0.45	0.052	4.2	0.78	0.049	5.2

2.10.8. Downstream Protective Values

In Suwannee Sound mechanistic models were applied to derive proposed DPVs for TN and TP shown in Table 2-89.

Table 2-89. DPVs for Suwannee Sound

Tributary	LSPC Model Watershed ID	USGS Reach Code	Estuary Segment	TN (mg/L)	TP (mg/L)
Suwannee River	70022	03110205000215	0817	0.88	0.166

2.10.9. References

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2.11. Springs Coast

2.11.1. Proposed Numeric Nutrient Criteria

Proposed numeric nutrient criteria for TN, TP, and chl-a in Springs Coast segments are summarized in Table 2-90.

Table 2-90. Proposed numeric nutrient criteria for Springs Coast segments

Segment Name	Segment Number	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
Anclote Offshore South	0803	0.29	0.008	2.6
Anclote River	0804	0.48	0.037	4.7
Anclote Offshore	0805	0.31	0.011	3.2
Pithlachascotee River	0806	0.50	0.022	2.4
Pithlachascotee Offshore	0807	0.32	0.011	2.5
Weeki Wachee River	0808	0.32	0.010	1.6
Weeki Wachee Offshore	0809	0.30	0.009	2.1
Chassahowitzka River	0810	0.32	0.010	0.7
Chassahowitzka River Offshore	0811	0.29	0.009	1.7
Crystal River	0812	0.35	0.013	1.3
Crystal-Homosassa Offshore	0813	0.36	0.013	2.1
Waccasassa River Offshore	0814	0.38	0.019	3.9
Cedar Keys	0815	0.32	0.019	4.1
Homosassa River	0833	0.47	0.032	1.9

2.11.2. General Characteristics

2.11.2.1. System Description

The Springs Coast region covers areas of the Waccasassa, Crystal (which covers most of the area designated by FDEP as the Springs Coast area), and Withlacoochee watersheds, extending from the Pithlachascotee River Basin north of Tampa Bay to the mouth of the Suwannee River.[62] The region includes large areas of marsh and wetland and borders the southern end of the Florida Big Bend Seagrass Beds Preserve, which boasts Florida's largest expanse of seagrass (Wolfe et al. 1990). Much of the Springs Coast shoreline is conserved by local, state, or federal entities. As a result of preservation efforts, salt marshes and mangrove forests remain intact, providing a buffer between uplands and estuarine waters (FDEP 2010a).

The Waccasassa watershed consists of the Waccasassa River watershed and surrounding small tributaries (Inwood 2009). The Waccasassa River is the main water body in the watershed and is approximately 28 mi (45 km) long. Estuarine portions of the river are also part of the watershed. While the Waccasassa River lacks major tributaries, numerous small tributaries contribute to the flow. Tannins and high acidity characterize the upper reaches of the river. In the river's middle reaches, groundwater input from springs and seepage dilute the acidic swamp water, creating a

[62] The information presented in this system description was compiled to summarize local information pertaining to the proposed numeric nutrient criteria for this estuary. For more information on EPA's process of delineating, segmenting, and deriving numeric nutrient criteria for this estuary, please see Section 1.3.

habitable environment for a variety of ecological communities. The lower reaches are tidally influenced and estuarine. The interaction between groundwater and surface water is important to the watershed's ecological health and aquatic resources (FDEP 2003). The Waccasassa watershed is 41 percent forested, 31 percent wetland, 12 percent urban, and 12 percent agricultural (Fry et al. 2011; SRWMD No date; SWFWMD 2007).[63]

The Withlacoochee watershed covers 2,100 mi^2 (5,400 km^2) and is dominated by the Withlacoochee River. Major water features in the watershed are the Withlacoochee River, Rainbow River, and Rainbow Springs. The Rainbow Springs system is the fourth largest spring in the state and the tenth largest freshwater spring worldwide. The Rainbow River is designated as an Aquatic Preserve, and Rainbow Springs is designated as a National Natural Landmark (FDEP 2011b). The Withlacoochee watershed is characterized by sandy ridges, which are very permeable and allow water to infiltrate and recharge the underlying Floridan aquifer (Amy H. Remley Foundation 2011). The Withlacoochee watershed is 28 percent forested or clear-cut, 27 percent agriculture land, 24 percent wetlands, and 17 percent urban areas (Fry et al. 2011; SRWMD No date; SWFWMD 2007).[64]

The Crystal-Pithlachascotee watershed is approximately 1,052 mi^2 (2,725 km^2), and the estuarine ecosystem covers approximately 153 mi^2 (396 km^2). The estuary's bays, rivers, salt marshes, seagrass meadows, oyster bars, and tidal flats encompass about 15 percent of the watershed. Major water features are the Crystal River, Kings Bay, Homosassa Springs, Chassahowitzka Springs, Weeki Wachee Spring, Anclote River, Pithlachascotee River, and associated coastal aquatic resources. The watershed includes several major spring complexes that occur because of the region's karst geology. Those springs are recharged primarily by rainfall, and all except for Weeki Wachee are affected by tidal flux. The northern part of the watershed is characterized by shallow waters, abundant freshwater flows, and low-energy shoreline—similar to an estuary (FDEP 2011a). The Crystal watershed is 41 percent urban areas, 22 percent forested, and 19 percent wetlands (Fry et al. 2011; SWFWMD 2007).[65]

2.11.2.2. Impaired Waters[66]

Three Class II and Class III marine WBIDs in the Waccasassa watershed have been listed for a nutrient-related parameter on Florida's CWA section 303(d) list approved by EPA. Of the three WBIDs, one is a Class II WBID, and two are Class III marine WBIDs. The Class II WBID is impaired for DO (WBID 1326). The two Class III marine WBIDs are impaired for nutrients and chl-a (WBIDs 3729 and 8037).

No Class II or Class III marine WBIDs with nutrient-related impairments are documented for the Withlacoochee watershed.

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[63] See Volume 1, Appendix C and its attachments for a summary of how land use was combined and for a detailed breakdown of land uses.
[64] See Volume 1, Appendix C and its attachments for a summary of how land use was combined and for a detailed breakdown of land uses.
[65] See Volume 1, Appendix C and its attachments for a summary of how land use was combined and for a detailed breakdown of land uses.
[66] For more information about the data source, see Volume 1, Appendix A.

Three Class III marine WBIDs in the Crystal watershed have been listed for a nutrient-related parameter on FDEP's CWA section 303(d) list approved by EPA. Of the three Class III marine WBIDs, two are impaired for DO (WBIDs 1345D and 1440) and one is impaired for nutrients, chl-a, and DO (WBID 1440A).[67]

No Class II or Class III marine WBIDs with nutrient-related TMDLs were documented within any of these regions.[68]

2.11.2.3. Water Quality

Mote Marine Laboratory conducted a 2-year study of water quality during 1984 and 1985 at river and estuary stations of the Waccasassa, Withlacoochee, Crystal, Weeki Wachee, and Aripeka rivers (Dixon 1986).

Mote Marine Laboratory determined that average DO levels were high throughout the water column, with minimum DO concentrations typically greater than 5.0 mg/L for sampled sites (along each river reach and in the estuary) (Dixon 1986). No hypoxic episodes are known to have occurred in the Springs Coast area (FDEP 2010a).

Mote Marine Laboratory found that the Waccasassa River had the highest chl-a concentrations, averaging 13.6 µg/L at the mouth of the river. Lowest chl-a values were observed on the Weeki Wachee River, where mean chl-a values were below 2.0 µg/L (Dixon 1986). FDEP analyzed data from the same study, which was available in the Legacy STOrage and RETrieval of Water-Related Data (STORET) database, and FDEP calculated the annual geometric mean of chl-a (uncorrected). The calculated chl-a values in Crystal River were 3.19 and 2.04 µg/L in 1984 and 1985, respectively. The calculated chl-a values in Weeki Wachee River were 1.78 and 0.62 µg/L in 1984 and 1985, respectively (FDEP 2010a). Increases in chlorophyll concentrations paralleled increases in TN and TP in Citrus, Hernando and Levy counties; the strongest correlation was with TP (Jacoby et al. 2009). For Cedar Keys, SRWMD found that the annual geometric mean for chl-a ranged from 2 to 14 µg/L between 1996 and 2006, with a general downward trend (FDEP 2010b).

In its 1984–1985 study, Mote Marine Laboratory measured water clarity offshore from the spring-fed rivers of Springs Coast and found that 20 percent of surface light reached a depth of 12.5 ft (3.8 m) on the Weeki Wachee River. Other parameters associated with water clarity, including turbidity, color, and TSS, were also measured in each river. Mean turbidity in the estuarine portions of the rivers during 1984 and 1985 ranged from 1.0 to 15.4 NTU, and mean

[67] The nutrient-related 303(d) list was a compilation of EPA-approved 303(d) listing information for nutrients, chl-a, and DO provided in three decision documents: September 2, 2009, Basin Groups 1, 2, and 5 EPA Decision Document (http://www.epa.gov/region4/water/tmdl/florida/documents/fl09303d_decisiondoc_090209.pdf);
May 13, 2010, Basin Group 3 EPA Decision Document
(http://www.epa.gov/region4/water/tmdl/florida/documents/fl303d_%20partialapproval_decision_docs051410.pdf);
and the December 21, 2010, Basin Group 4 EPA Decision Document
(http://www.epa.gov/region4/water/tmdl/florida/documents/group_4_final_dec_doc_and_partial_app_letter_12_21_10.pdf).

[68] TMDLs were identified in February 2011 by compiling nutrient-related draft/final TMDLs from the following three sources:
FDEP TMDL website (http://www.dep.state.fl.us/water/tmdl/index.htm)
EPA Region 4 website (http://www.epa.gov/region4/water/tmdl/florida/index.html)
EPA National WATERS expert query tool (http://www.epa.gov/waters/tmdl/expert_query.html).

color ranged from 7 to 45 PCU. TSS was measured at surface, mid, and bottom depths. Mean surface TSS in estuarine portions of the rivers ranged from 3 to 22 mg/L, mean mid-depth TSS ranged from 3 to 36 mg/L, and mean bottom TSS ranged from 4 to 34 mg/L. The highest value for each of these water clarity parameters occurred at the sampling station closest to the mouth of the Waccasassa River (Dixon 1986).

The annual geometric mean TN concentrations for 1984 and 1985 ranged from 0.472 to 0.504 mg/L in the Crystal River and was 0.611 mg/L for both years in the Weeki Wachee River. A strong relationship between nitrogen and salinity was not observed (FDEP 2010a). SRWMD found the annual geometric mean TN in Cedar Keys ranged from 0.4 to 1.2 mg/L between 1996 and 2006, with the lowest TN in 1999 and the highest in 2004 and 2005 (FDEP 2010b).

Elevated nitrate levels in the basin's springs are attributed to pollutant sources in their springsheds. Primary nitrate sources are inorganic in nature, such as fertilizers (FDEP 2008). Harrington et al. (2010) found that in the springshed of Weeki Wachee Spring, primary sources of nitrates are from septic systems and fertilizers applied to lawns and golf courses, which come from a large area of medium-density residential development (Harrington et al. 2010).

The natural abundance of phosphorus varies across the state (including the Springs Coast area), as do background groundwater concentrations (Harrington et al. 2010). The Florida Springs Initiative reported median phosphate concentrations found in groundwater for the surficial (0.116 mg/L) and Floridan (0.027 mg/L) aquifer systems for 1985–2006. Median phosphate levels reported for the springs of the Springs Coast area are all between 0.006 and 0.030 mg/L for the eight basin groups in the state that have springs (Harrington et al. 2010). Phosphate levels in the Springs Coast planning units are at or near historical background concentrations, with the exception of two springs in the Anclote River Planning Unit, which had a median phosphate value of 0.07 mg/L (FDEP 2008). SRWMD found that the annual geometric mean TP for Cedar Keys ranged from 0.04 to 0.09 mg/L between 1996 and 2006 (FDEP 2010b).

Annual load estimates for TN and TP in the Homosassa and Weeki Wachee rivers were calculated on the basis of annual mean nutrient concentrations from bimonthly sampling at one Southwest Florida Water Management District (SWFWMD) site on each river and mean annual discharge at U.S. Geological Survey gauge stations. The annual TN load for the Homosassa River was 215,526 lb/yr (97,761 kg/yr), and the annual TN load for the Weeki Wachee River was 198,674 lb/yr (90,117 kg/yr) (FDEP 2010a). The annual TP load for the Homosassa River was 12,923 lb/yr (5,866 kg/yr), and the annual TP load for the Weeki Wachee River was 3,036 lb/yr (1,377 kg/yr) (FDEP 2010a).

2.11.2.4. Biological Characteristics

The Springs Coast estuarine area provides essential habitat for many fish and wildlife species, including nursery and juvenile habitats for many recreational and commercial fish species. Seagrass beds, which provide a breeding habitat for many species, cover almost the entire nearshore area along the watershed's northern portion. Extensive oyster reefs are also present (FDEP 2011a).

Additionally, the Withlacoochee watershed ecosystem supports nearly 500 vertebrate species, including freshwater and saltwater fish, amphibians, reptiles, birds, and mammals (FDEP 2011b). The Crystal River provides critical habitat for manatees. The Crystal River herd composes about 25 percent of the U.S. manatee population (FDEP 2011a).

The wetlands of the Springs Coast act as a nursery and a food source, supporting much of the gulf fishery (Wolfe et al. 1990). Vast salt marshes in the area extend up to 10 mi (16 km) inland. However, well-drained soils in a karst landscape contribute to the low percentage of wetlands and the higher percentage of upland forests in the springshed areas as compared to other watersheds. The Weeki Wachee Preserve holds the southernmost coastal hardwood hammock in western Florida (FDEP 2010a). Between 2006 and 2009, seagrass coverage declined around Cedar Keys. Reduction of water clarity caused by increased nutrients, phytoplankton, and turbidity is a stressor to the seagrasses of this area (FFWCC 2011).

The coastline's expansive SAV community, including seagrasses and associated macroalgae, provides habitat for fish, manatees, sea turtles, and other wildlife. Along the Springs Coast, SAV is the most sensitive marine community to nutrient pollution and associated light reduction (FDEP 2010a). As of 2007, seagrass covers 592 mi^2 (1,534 km^2 [379,010 ac]) of the Springs Coast region. Compared to 1999 seagrass data, the 2007 coverage and species composition appear to be stable. However, some conversion of continuous beds to patchy seagrass beds has been noted. Stressors to the Springs Coast's seagrass population include nutrients, phytoplankton, and turbidity (FFWCC 2011). No HABs have been documented in the Springs Coast area (FDEP 2010a).

Cedar Keys has been studied by the FFWCC, and monitoring has been ongoing since 1996 by the Commission's Fisheries-Independent Monitoring program (FWRI 2009). Sampling efforts in 2008 showed that bay anchovy (*Anchoa mitchili*) represented over 54 percent of the total catch with more than 95,951 fish caught (FWRI 2009).

For a more detailed summary of this water body, see Volume 1, Appendix A.

2.11.3. Data Used

Several data sources specific to Springs Coast were used in addition to those sources described in Section 1.4.3, as summarized in Table 2-91.

Table 2-91. Data sources specific to Springs Coast models

Data	Source	Location Used
Springs discharge and water quality data	Florida Department of Environmental Protection (FDEP 2004, 2009; Hornsby and Ceryak 2000)	Apalachicola, Suwannee, Econfina, Waccasassa, Crystal, and Withlacoochee watershed models
FDEP Level III Florida Land Use	Suwannee River Water Management District (SRWMD No date)	Apalachee, Econfina, Suwannee, and Waccasassa watershed models
Climate data	Florida State Climate Center (FSCC 2009) and EarthInfo (EarthInfo 2009)	All watershed models
Municipal and industrial point sources	Georgia Environmental Protection Division (GAEPD No date)	Suwannee and Apalachee watershed models
Water quality data	Georgia Environmental Protection Division (GAEPD 2008)	Suwannee and Apalachee watershed models
Georgia Land Use Trends land use	Natural Resources Spatial Analysis Lab (NARSAL 2008)	Suwannee and Apalachee watershed models
FDEP Level III Florida Land Use	Southwest Florida Water Management District (SWFWMD 2007)	Waccasassa, Crystal, and Withlacoochee watershed models

2.11.4. Segmentation

Springs Coast was divided into segments on the basis of the existing WBIDs. In some cases where there was homogeneity in seagrass distribution, two or more adjoining WBIDs were combined into one segment. The segments were limited to a distance of 3 nautical miles from the shoreline, where the estuary numeric nutrient criteria would be applicable. Figure 2-34 shows the resulting segments for Springs Coast.

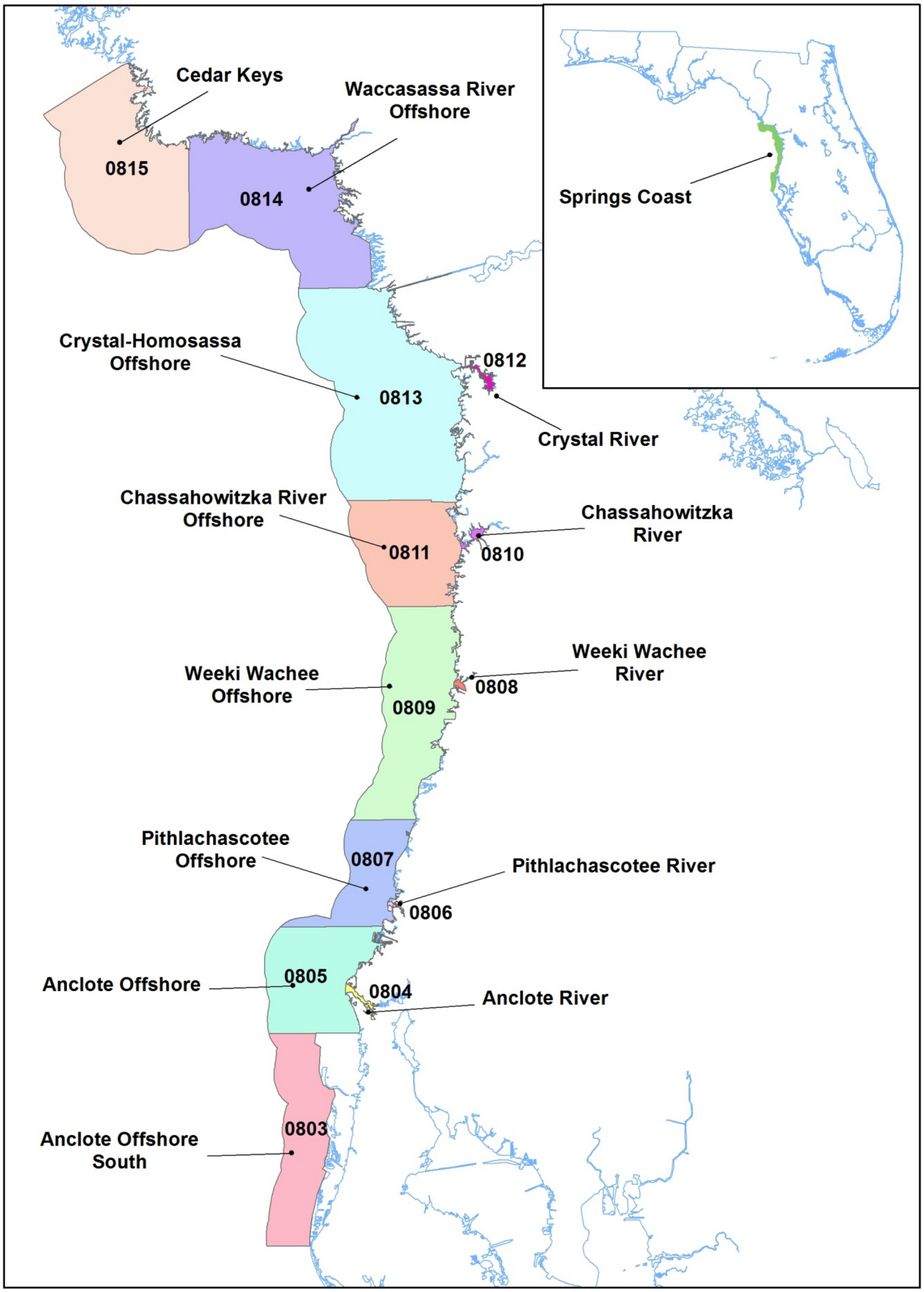

Figure 2-34. Results of Springs Coast segmentation

2.11.5. Water Quality Targets

2.11.5.1. Seagrass Depth and Water Clarity Targets

Seagrass depth of colonization (measured as Z_c) and water clarity (measured as K_d) targets were established for two of the Springs Coast Estuary segments by averaging the depth of colonization targets of WBIDs in each estuary segment, as shown in Table 2-92 (SWFWMD 2012).

Table 2-92. Springs Coast seagrass depth and water clarity targets by segment

Estuary Segment	Depth of Colonization (Z_c) Target (m)	Light Attenuation Coefficient (K_d) Target (1/m)
0803	No target	-
0804	No target	-
0805	1.8	0.9
0806	1.6	1.0
0807	No target	-
0808	No target	-
0809	No target	-
0810	No target	-
0811	No target	-
0812	No target	-
0813	No target	-
0814	No target	-
0815	No target	-
0833	No target	-

2.11.5.2. Chlorophyll a Target

To prevent nuisance algal blooms and protect the estuary's designated uses, chl-a levels must not exceed 20 µg/L more than 10 percent of the time. The rationale for that target is provided in Section 1.2.2.

2.11.5.3. Dissolved Oxygen Targets

On the basis of the rationale that sufficient DO is necessary to protect aquatic life, as described in Section 1.2.3, the following DO targets were established:

- Minimum allowable DO of 4.0 mg/L as a water column average for each estuary segment 90 percent of the time over the simulation's time span
- Daily average DO of 5.0 mg/L as a water column average for each estuary segment 90 percent of the time over the simulation's time span
- Minimum 3-hour average DO of 1.5 mg/L in the bottom two layers for each estuary segment over the simulation's time span

2.11.6. Results of Analyses

2.11.6.1. Mechanistic Model Analysis

Average load contributions to Springs Coast from the Waccasassa Withlacoochee, and Crystal watersheds are shown in Table 2-93, Table 2-94, and Table 2-95.

Table 2-93. Average load contributions from the Waccasassa watershed

	TN Load1 (kg/d; Mean + se)			TP Load1 (kg/d; Mean + se)		
Year	Existing[2]	Background[3]	Anthropogenic[4]	Existing[2]	Background[3]	Anthropogenic[4]
2002	1412 ± 84	867 ± 60	546 ± 25	102 ± 7	55 ± 5	47 ± 3
2003	2282 ± 165	1425 ± 122	858 ± 45	158 ± 13	83 ± 9	75 ± 5
2004	2497 ± 214	1599 ± 151	899 ± 64	154 ± 14	85 ± 10	69 ± 5
2005	1834 ± 88	1073 ± 56	762 ± 33	132 ± 6	65 ± 4	67 ± 3
2006	711 ± 48	418 ± 32	294 ± 16	47 ± 2	24 ± 1	22 ± 1
2007	723 ± 66	421 ± 45	302 ± 22	48 ± 3	25 ± 2	23 ± 1
2008	658 ± 65	363 ± 41	295 ± 25	47 ± 4	24 ± 2	23 ± 2
2009	998 ± 95	584 ± 65	414 ± 31	86 ± 8	43 ± 4	43 ± 4

[1] Annual average daily load from all terminal reaches
[2] Annual average daily load from all terminal reaches computed from existing model scenario
[3] Annual average daily load from all terminal reaches computed from background model scenario (remove anthropogenic sources/land use; use existing hydrology)
[4] Anthropogenic load computed as difference between existing and background load

Table 2-94. Average load contributions from the Withlacoochee watershed

	TN Load[1] (kg/d; Mean + se)			TP Load[1] (kg/d; Mean + se)		
Year	Existing[2]	Background[3]	Anthropogenic[4]	Existing[2]	Background[3]	Anthropogenic[4]
2002	3770 ± 207	1801 ± 99	1969 ± 109	224 ± 8	141 ± 6	83 ± 2
2003	5895 ± 330	2726 ± 159	3169 ± 172	240 ± 8	141 ± 6	99 ± 3
2004	5232 ± 545	2408 ± 248	2824 ± 298	230 ± 11	144 ± 8	86 ± 4
2005	3677 ± 198	1710 ± 94	1966 ± 105	206 ± 8	135 ± 6	71 ± 2
2006	1332 ± 76	691 ± 39	641 ± 39	159 ± 6	107 ± 4	52 ± 2
2007	1729 ± 86	895 ± 45	834 ± 42	180 ± 7	123 ± 5	57 ± 2
2008	2016 ± 131	1013 ± 65	1003 ± 67	161 ± 6	105 ± 4	55 ± 2
2009	1613 ± 81	839 ± 43	773 ± 39	184 ± 6	125 ± 5	60 ± 2

[1] Annual average daily load from all terminal reaches
[2] Annual average daily load from all terminal reaches computed from existing model scenario
[3] Annual average daily load from all terminal reaches computed from background model scenario (remove anthropogenic sources/land use; use existing hydrology)
[4] Anthropogenic load computed as difference between existing and background load

Table 2-95. Average load contributions from the Crystal watershed

Year	TN Load[1] (kg/d; Mean + se)			TP Load[1] (kg/d; Mean + se)		
	Existing[2]	Background[3]	Anthropogenic[4]	Existing[2]	Background[3]	Anthropogenic[4]
2002	4815 ± 327	2373 ± 184	2442 ± 147	210 ± 17	115 ± 12	95 ± 6
2003	5264 ± 493	2594 ± 297	2670 ± 201	208 ± 22	114 ± 16	94 ± 6
2004	4503 ± 716	2382 ± 458	2122 ± 263	199 ± 35	119 ± 28	80 ± 8
2005	3814 ± 241	1917 ± 143	1897 ± 101	172 ± 13	96 ± 9	75 ± 4
2006	3109 ± 296	1625 ± 178	1484 ± 121	141 ± 16	84 ± 12	57 ± 4
2007	1552 ± 140	860 ± 83	693 ± 60	78 ± 8	47 ± 5	31 ± 3
2008	2372 ± 194	1242 ± 116	1131 ± 82	114 ± 10	67 ± 7	47 ± 3
2009	2990 ± 273	1601 ± 164	1389 ± 111	153 ± 14	91 ± 10	61 ± 5

[1] Annual average daily load from all terminal reaches
[2] Annual average daily load from all terminal reaches computed from existing model scenario
[3] Annual average daily load from all terminal reaches computed from background model scenario (remove anthropogenic sources/land use; use existing hydrology)
[4] Anthropogenic load computed as difference between existing and background load

For Springs Coast, the chl-a target was met for each segment on the basis of 2002–2009 nutrient loads. DO targets were not met for segments 0804, 0806, 0808, 0810, 0812, and 0833. Those segments could not meet the DO targets using either the 2002–2009 nutrient loads scenario or the non-anthropogenic nutrient scenario. The light attenuation coefficient target was not met for segment 0806 under either the 2002–2009 nutrient loads or the non-anthropogenic nutrient scenario. Table 2-96 identifies which targets were met under 2002–2009 nutrient loads and which targets were not met.

An evaluation of model sensitivity to the water quality targets applied revealed that light and DO targets were insensitive to changes in nutrients in Springs Coast.

Table 2-96. Water quality targets met for Springs Coast based on mechanistic modeling

Segment	DO	Chl-a	K_d
0803	Yes	Yes	No target
0804	No	Yes	No target
0805	Yes	Yes	Yes
0806	No	Yes	No
0807	Yes	Yes	No target
0808	No	Yes	No target
0809	Yes	Yes	No target
0810	No	Yes	No target
0811	Yes	Yes	No target
0812	No	Yes	No target
0813	Yes	Yes	No target
0814	Yes	Yes	No target
0815	Yes	Yes	No target
0833	No	Yes	No target

A summary of candidate criteria based on the 2002–2009 nutrient loads for Springs Coast Estuary segments is given in Table 2-97.

Table 2-97. Summary of candidate criteria for Springs Coast derived from mechanistic modeling

Segment	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
0803	0.29	0.008	2.6
0804	0.48	0.037	4.7
0805	0.31	0.011	3.2
0806	0.50	0.022	2.4
0807	0.32	0.011	2.5
0808	0.32	0.010	1.6
0809	0.30	0.009	2.1
0810	0.32	0.010	0.7
0811	0.29	0.009	1.7
0812	0.35	0.013	1.3
0813	0.36	0.013	2.1
0814	0.38	0.019	3.9
0815	0.32	0.019	4.1
0833	0.47	0.032	1.9

2.11.6.2. Statistical Model Analysis

No water clarity targets were available where data were sufficient to estimate relationships between water clarity and chl-a. Data were sufficient to estimate the relationship between bloom frequency and annual geometric mean chl-a concentration in segments 0812, 0813, and 0815 (Crystal and Waccasassa Rivers). The frequency of algal blooms did increase with chl-a, and a candidate chl-a criterion was calculated as the chl-a concentration at which predicted bloom frequency was 10 percent. In segment 0813, the calculated chl-a criterion was greater than the upper bound of observed annual geometric mean chl-a concentrations. Hence, the candidate chl-a criterion was set to the upper bound of observed values for this segment (Figure 2-35).

Relationships between TN, TP, and chl-a were estimated to derive candidate TN and TP concentrations associated with chl-a. In segment 0812, calculated criteria for both TN and TP were greater than the upper bound of the observed TN and TP values, so candidate criteria were set at the upper bound. In segment 0813, a TP criterion was calculated such that predicted annual geometric mean chl-a concentrations were equivalent to the candidate criteria. However, the calculated TN criterion in this segment was greater than the upper bound of observed values, and the criterion was set at the upper bound. In segment 0815, chl-a increased with increasing TP, but the calculated criterion was less than the lower bound of observed values, so the candidate criterion was set at the lower bound. In this segment, chl-a exhibited no relationship with TN, and no criterion was computed (Table 2-98).

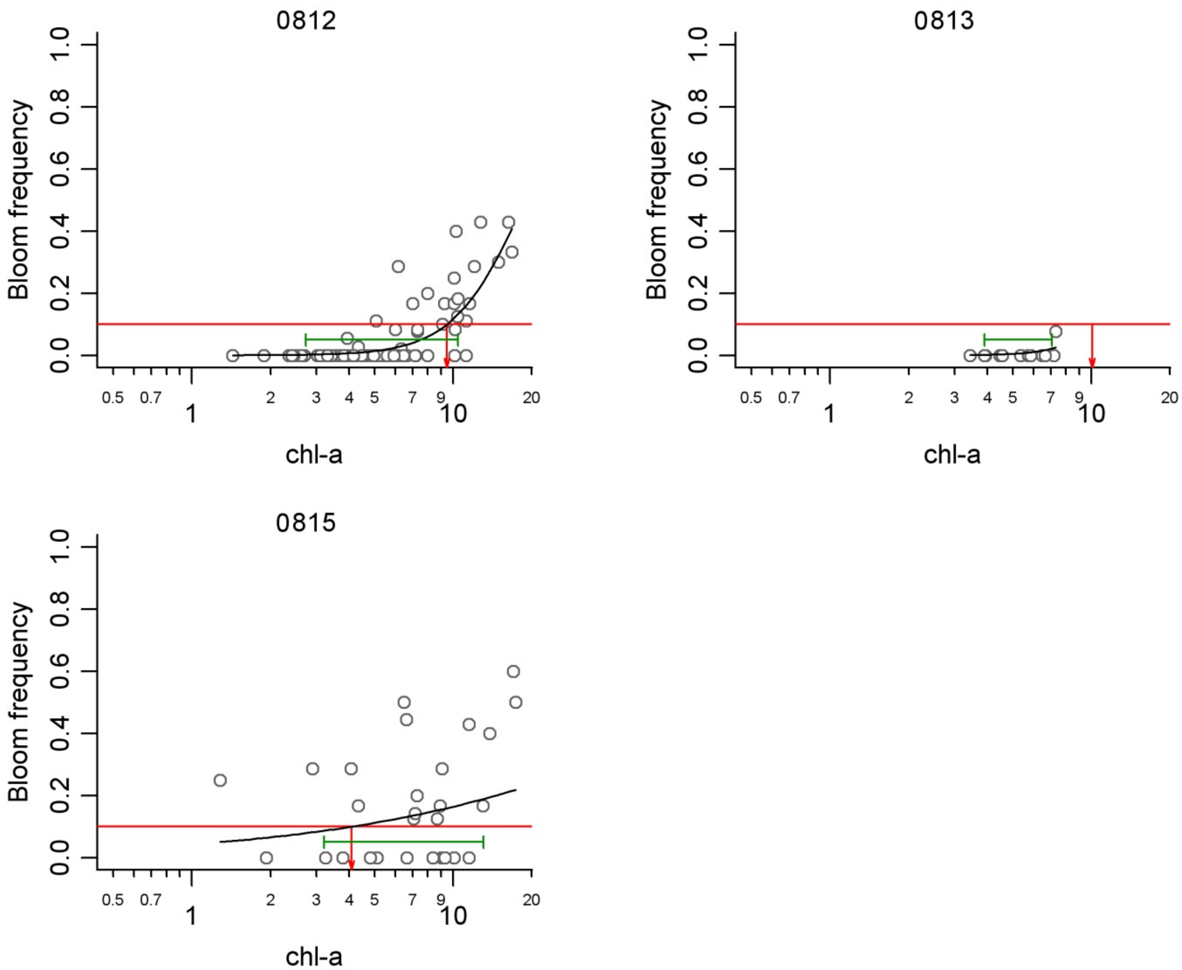

Figure 2-35. Relationships between phytoplankton blooms and annual geometric mean chl-a in Crystal River (0812 and 0813) and Waccasassa River (0815). Open circles: observed annual geometric mean chl-a and bloom frequency; red horizontal line: targeted bloom frequency of 10 percent; red vertical arrow: chl-a concentration corresponding to targeted bloom frequency; green line segment: 5th to 95th percentile range of observed data.

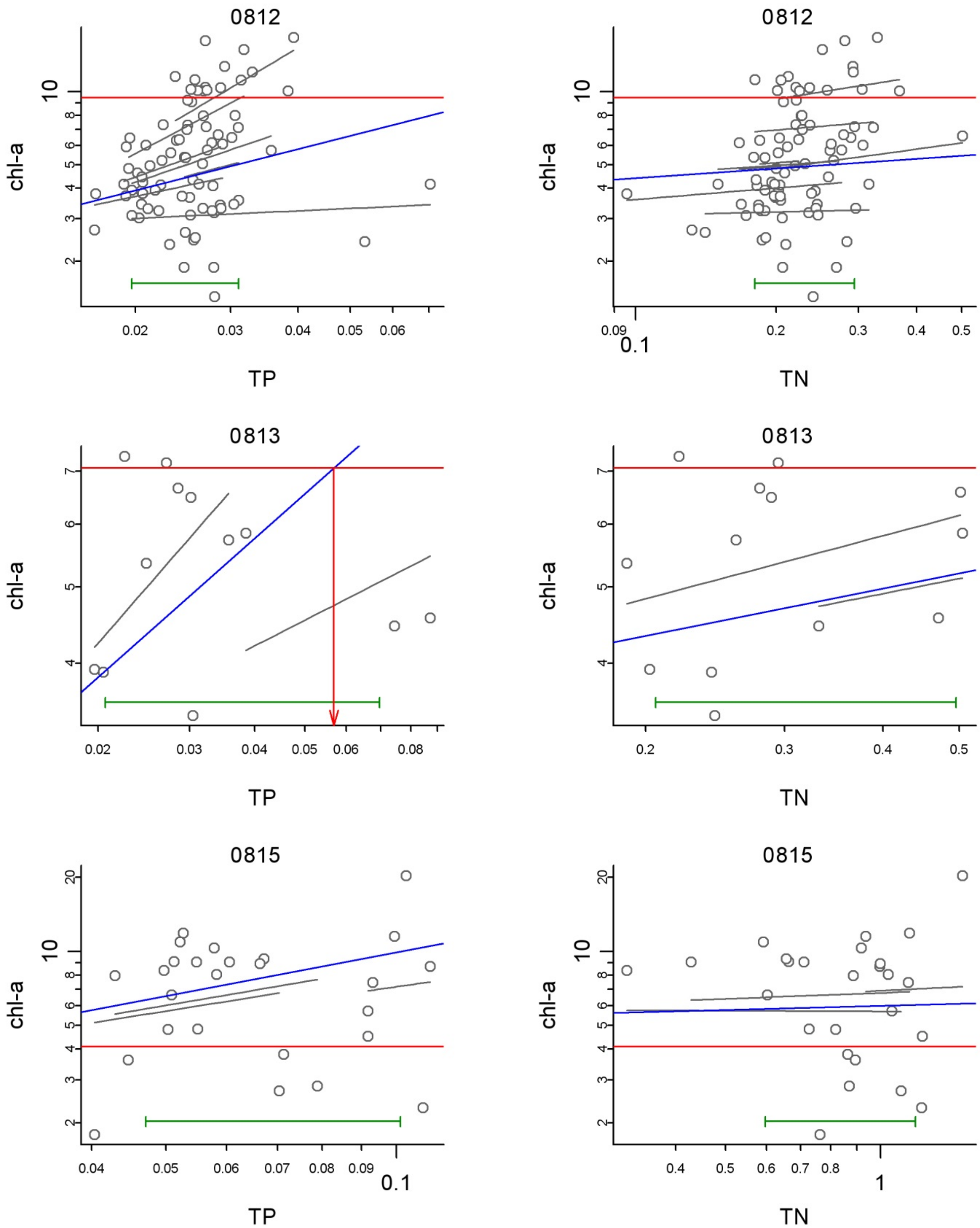

Figure 2-36. Relationships between TN, TP, and chl-a in Crystal River and Waccasassa River. Open circles: observed annual average values of TN, TP, and chl-a, red horizontal line: chl-a criterion, red vertical arrow: TN and TP criterion associated with chl-a criterion, blue line: estimated segment-wide relationship between TN, TP, and chl-a, grey lines: estimated station-specific relationships between TN, TP, and chl-a, green line segment: 5th to 95th percentile range of observed annual geometric mean TN and TP concentrations.

Table 2-98. Summary of TN, TP, and chl-a criteria derived by statistical analysis for Crystal River and Waccasassa River. Criteria values with asterisks represent either the upper or lower bound of observed values.

Segment	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
0812	0.29*	0.031*	9.5
0813	0.50*	0.057	7.1*
0815	-	0.047*	4.1

2.11.7. Application of Analyses for Proposed Numeric Nutrient Criteria

EPA evaluated various lines of evidence for deriving numeric nutrient criteria in Anclote River/Anclote Offshore. There were insufficient data in Anclote River/Anclote Offshore to derive the proposed criteria using statistical models. As a result, EPA derived the proposed numeric nutrient criteria for Anclote River/Anclote Offshore using the mechanistic modeling results.

EPA evaluated three endpoints in the mechanistic modeling approach for Anclote River/Anclote Offshore: (1) water clarity protective of historic depth of seagrasses, (2) chl-a concentrations associated with balanced phytoplankton biomass, and (3) DO concentrations sufficient to maintain aquatic life. EPA found that the DO target was not met under the calibrated 2002–2009 nutrient loads and was insensitive to changes in nutrients. Therefore, the DO endpoint was not used in Anclote River/Anclote Offshore. However, the water clarity and chl-a targets were met under the calibrated 2002–2009 nutrient loads and were sensitive to changes in nutrients. The proposed criteria were derived to be protective of water clarity and chl-a concentrations. The values under mechanistic modeling represent the 90th percentile annual geometric mean nutrient concentrations from the 2002–2009 modeled nutrient loads.

Proposed numeric nutrient criteria for TN, TP, and chl-a in Anclote River and offshore segments are summarized in Table 2-99.

Table 2-99. Proposed and candidate numeric nutrient criteria for Anclote River and offshore segments

		Proposed Criteria			Mechanistic Modeling		
SEGMENT	SEGMENT ID	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
Anclote Offshore South	0803	0.29	0.008	2.6	0.29	0.008	2.6
Anclote River	0804	0.48	0.037	4.7	0.48	0.037	4.7
Anclote Offshore	0805	0.31	0.011	3.2	0.31	0.011	3.2

EPA evaluated various lines of evidence for deriving numeric nutrient criteria in Pithlachascotee River/Pithlachascotee Offshore. There were insufficient data in this area to derive the proposed criteria using statistical models. As a result, EPA derived the proposed numeric nutrient criteria for Pithlachascotee River/Pithlachascotee Offshore using the mechanistic modeling results.

EPA evaluated three endpoints in the mechanistic modeling approach for Pithlachascotee River/Pithlachascotee Offshore: (1) water clarity protective of historic depth of seagrasses, (2) chl-a concentrations associated with balanced phytoplankton biomass, and (3) DO concentrations sufficient to maintain aquatic life. EPA found the water clarity and DO endpoints were not met in all segments under the calibrated 2002–2009 nutrient loads and were insensitive to changes in nutrients loads. As a result, the water clarity and DO endpoints were not used in Pithlachascotee River/Pithlachascotee Offshore. The chl-a target was met and was shown to be sensitive to changes in nutrients. The proposed criteria were derived to be protective of chl-a concentrations. The values under mechanistic modeling represent the 90th percentile annual geometric mean nutrient concentrations from the 2002–2009 modeled nutrient loads.

Proposed numeric nutrient criteria for TN, TP, and chl-a in Pithlachascotee River and offshore segments are summarized in Table 2-100.

Table 2-100. Proposed and candidate numeric nutrient criteria for Pithlachascotee River and offshore segments

		Proposed Criteria			Mechanistic Modeling		
SEGMENT	SEGMENT ID	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
Pithlachascotee River	0806	0.50	0.022	2.4	0.50	0.022	2.4
Pithlachascotee Offshore	0807	0.32	0.011	2.5	0.32	0.011	2.5

EPA evaluated various lines of evidence for deriving numeric nutrient criteria in Weeki Wachee/Weeki Wachee Offshore. There were insufficient data in Weeki Wachee/Weeki Wachee Offshore to derive the proposed criteria using statistical models. As a result, EPA derived the proposed numeric nutrient criteria for Weeki Wachee/Weeki Wachee Offshore using the mechanistic modeling results.

Because depth of colonization targets were not available in Weeki Wachee/Weeki Wachee Offshore EPA evaluated two endpoints in the mechanistic modeling approach: (1) chl-a concentrations associated with balanced phytoplankton biomass and (2) DO concentrations sufficient to maintain aquatic life. EPA found the DO endpoint was not met in all segments under the calibrated 2002–2009 nutrient loads and was insensitive to changes in nutrients loads. As a result, the DO endpoint was not used in Weeki Wachee/Weeki Wachee Offshore. The chl-a target was met and was shown to be sensitive to changes in nutrients. The proposed criteria were derived to be protective of chl-a concentrations. The values under mechanistic modeling represent the 90th percentile annual geometric mean nutrient concentrations from the 2002–2009 modeled nutrient loads.

Proposed numeric nutrient criteria for TN, TP, and chl-a in Weeki Wachee segments are summarized in Table 2-101.

Table 2-101. Proposed and candidate numeric nutrient criteria for Weeki Wachee segments

		Proposed Criteria			Mechanistic Modeling		
SEGMENT	SEGMENT ID	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
Weeki Wachee River	0808	0.32	0.010	1.6	0.32	0.010	1.6
Weeki Wachee Offshore	0809	0.30	0.009	2.1	0.30	0.009	2.1

EPA evaluated various lines of evidence for deriving numeric nutrient criteria in Chassahowitzka River/Chassahowitzka Offshore. There were insufficient data in this area to derive the proposed criteria using statistical models. As a result, EPA derived the proposed numeric nutrient criteria for Chassahowitzka River/Chassahowitzka Offshore using the mechanistic modeling results.

Because depth of colonization targets were not available in Chassahowitzka River/Chassahowitzka Offshore, EPA evaluated two endpoints in the mechanistic modeling approach: (1) chl-a concentrations associated with balanced phytoplankton biomass and (2) DO concentrations sufficient to maintain aquatic life. EPA found the DO endpoint was not met in all segments under the calibrated 2002–2009 nutrient loads and was insensitive to changes in nutrients loads. As a result, the DO endpoint was not used in Chassahowitzka River/Chassahowitzka Offshore. The chl-a target was met and was shown to be sensitive to changes in nutrients. The proposed criteria were derived to be protective of chl-a concentrations. The values under mechanistic modeling represent the 90th percentile annual geometric mean nutrient concentrations from the 2002–2009 modeled nutrient loads.

Proposed numeric nutrient criteria for TN, TP, and chl-a in Chassahowitzka segments are summarized in Table 2-102.

Table 2-102. Proposed and candidate numeric nutrient criteria for Chassahowitzka segments

		Proposed Criteria			Mechanistic Modeling		
SEGMENT	SEGMENT ID	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
Chassahowitzka River	0810	0.32	0.010	0.7	0.32	0.010	0.7
Chassahowitzka River Offshore	0811	0.29	0.009	1.7	0.29	0.009	1.7

EPA evaluated various lines of evidence for deriving numeric nutrient criteria in Crystal River/Homosassa. There were insufficient data in this area to derive the proposed criteria using statistical models. As a result, EPA derived the proposed numeric nutrient criteria for Crystal River/Homosassa using the mechanistic modeling results.

Because water clarity targets were not available in Crystal River/Homosassa, EPA evaluated two endpoints in the mechanistic modeling approach: (1) chl-a concentrations associated with balanced phytoplankton biomass and (2) DO concentrations sufficient to maintain aquatic life. EPA found the DO endpoint was not met in all segments under the calibrated 2002–2009 nutrient loads and was insensitive to changes in nutrients loads. As a result, the DO endpoint was not used in Crystal River/Homosassa. The chl-a target was met and was shown to be sensitive to changes in nutrients. The proposed criteria were derived to be protective of chl-a concentrations. The values under mechanistic modeling represent the 90th percentile annual geometric mean nutrient concentrations from the 2002–2009 modeled nutrient loads.

Proposed numeric nutrient criteria for TN, TP, and chl-a in Crystal River and Homosassa segments are summarized in Table 2-103.

Table 2-103. Proposed and candidate numeric nutrient criteria for Crystal River and Homosassa segments

		Proposed Criteria			Mechanistic Modeling			Statistical Modeling		
SEGMENT	SEGMENT ID	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
Crystal River	0812	0.35	0.013	1.3	0.35	0.013	1.3	0.29	0.031	9.5
Crystal-Homosassa Offshore	0813	0.36	0.013	2.1	0.36	0.013	2.1	0.50	0.057	7.1
Homosassa River	0833	0.47	0.032	1.9	0.47	0.032	1.9	-	-	-

EPA evaluated various lines of evidence for deriving numeric nutrient criteria Waccasassa River Offshore/Cedar Keys. There were insufficient data in this area to derive the proposed criteria using statistical models. As a result, EPA derived the proposed numeric nutrient criteria for Waccasassa River Offshore/Cedar Keys using the mechanistic modeling results.

Because depth of colonization targets were not available in Waccasassa River Offshore/Cedar Keys, EPA evaluated two endpoints in the mechanistic modeling approach: (1) chl-a concentrations associated with balanced phytoplankton biomass and (2) DO concentrations sufficient to maintain aquatic life. EPA found that both of the endpoints were met under the calibrated 2002–2009 nutrient loads and sensitive to changes in nutrients. The proposed criteria were derived to be protective of both. The values under mechanistic modeling represent the 90th percentile annual geometric mean nutrient concentrations from the 2002–2009 modeled nutrient loads.

Proposed numeric nutrient criteria for TN, TP, and chl-a in Waccasassa River Offshore/Cedar Keys segments are summarized in Table 2-104.

Table 2-104. Proposed and candidate numeric nutrient criteria for Waccasassa River Offshore segments

		Proposed Criteria			Mechanistic Modeling			Statistical Modeling		
SEGMENT	SEGMENT ID	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
Waccasassa River Offshore	0814	0.38	0.019	3.9	0.38	0.019	3.9	-	-	-
Cedar Keys	0815	0.32	0.019	4.1	0.32	0.019	4.1	-	0.047	4.1

2.11.8. Downstream Protective Values

In Springs Coast mechanistic models were applied to derive proposed DPVs for TN and TP shown in Table 2-105.

Table 2-105. Proposed DPVs for Springs Coast

Tributary[a]	LSPC Model Watershed ID	USGS Reach Code	Estuary Segment	TN (mg/L)	TP (mg/L)
Giger Creek	110001	03110101000176	0815	0.66	0.019
Dry Creek	110007	03110101000292	0814	0.87	0.057
	110008	03110101005195	0814	0.94	0.037
	110010	03110101000447	0814	0.58	0.030
East Griffin Creek	110015	03110101001164	0814	0.88	0.034
Turtle Creek	110016	03110101000413	0814	0.72	0.032
Spring Run	110023	03110101000429	0814	0.67	0.071
	110026	03110101003844	0814	1.91	0.072
Bird Creek	110027	03110101000298	0814	1.09	0.026
Waccasassa River	110029	03110101000465	0814	0.63	0.059
Withlacoochee River	150032	03100208000478	0813	0.90	0.093
	140001	03100207000199	0813	0.05	0.002
Kings Creek	140002	03100207000843	0812	0.17	0.002
	140003	03100207000850	0813	0.79	0.026
Halls River	140004	03100207003282	0833	0.35	0.017
	140005	03100207000179	0813	0.60	0.014
Chassahowitzka River	140006	03100207001280	0810	0.30	0.009
	140007	03100207001265	0811	0.74	0.020
Weeki Wachee River	140008	03100207001575	0808	0.24	0.004
Jenkins Creek	140009	03100207003230	0809	0.81	0.021
Indian Creek	140010	03100207001021	0809	0.85	0.037
	140011	03100207001046	0809	0.70	0.017
Double Hammock Creek	140012	03100207016331	0807	1.12	0.026
Pithlachascotee River	140013	03100207000054	0807	0.65	0.027
Anclote River	140020	03100207000022	0804	0.30	0.016
	140021	03100207001082	0805	1.15	0.031

[a] Tributary names left blank are unnamed

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2.12. Clearwater Harbor/St. Joseph Sound

2.12.1. Proposed Numeric Nutrient Criteria

Newly-approved State water quality standards apply to Clearwater Harbor/St. Joseph Sound (Subsection 62-302.352(a), F.A.C.).

2.12.2. Downstream Protective Values

Using geographic information systems, EPA integrated a waterway coverage with the estuary segments in Clearwater Harbor/St. Joseph Sound and identified the pour point as the location where the waterway intersected with the estuary segment. In Clearwater Harbor/St. Joseph Sound, EPA applied Florida's approved numeric criteria (Table 2-106) from the Florida downstream receiving segment as the proposed DPVs for TN and TP shown in Table 2-107.

Table 2-106. Newly-approved State water quality standards for Clearwater Harbor/St. Joseph Sound

Estuary Segment	TN (mg/L)	TP (mg/L)
(a)1 St. Joseph Sound	0.66	0.05
(a)2 Clearwater North	0.61	0.05
(a)3 Clearwater South	0.58	0.06

Table 2-107. Proposed DPVs for Clearwater Harbor/St. Joseph Sound

Estuary Segment	Tributary	USGS Reach Code	TN (mg/L)	TP (mg/L)
(a)2 Clearwater North	Curlew Creek	03100207001148	0.61	0.05
(a)3 Clearwater South	Mckay Creek	03100207001160	0.58	0.06

2.13. Tampa Bay

2.13.1. Proposed Numeric Nutrient Criteria

Newly-approved State water quality standards apply to Tampa Bay (Subsection 62-302.352(b), F.A.C.).

2.13.2. Downstream Protective Values

Using geographic information systems, EPA integrated a waterway coverage with the estuary segments in Tampa Bay and identified the pour point as the location where the waterway intersected with the estuary segment. In Tampa Bay, EPA applied Florida's approved numeric criteria (Table 2-108) from the Florida downstream receiving segment as the proposed DPVs for TN and TP shown in Table 2-109.

Table 2-108. Newly-approved State water quality standards for Tampa Bay

Estuary Segment	TN (tons/million m^3 water)	TP (tons/million m^3 water)
(b)1 Old Tampa Bay	1.08	0.23
(b)2 Hillsborough Bay	1.62	1.28
(b)3 Middle Tampa Bay	1.24	0.24
(b)4 Lower Tampa Bay	0.97	0.14
(b)5 Boca Ciega North	1.54	0.18
(b)6 Boca Ciega South	0.97	0.06
(b)7 Terra Ceia Bay	1.10	0.14
(b)8 Manatee River Estuary	1.80	0.37

Table 2-109. Proposed DPVs for Tampa Bay

Estuary Segment	Tributary[a]	USGS Reach Code	TN (tons/million m^3 water)	TP (tons/million m^3 water)
(b)1 Old Tampa Bay		03100206011290	1.08	0.23
(b)1 Old Tampa Bay		03100206000266	1.08	0.23
(b)1 Old Tampa Bay		03100206000300	1.08	0.23
(b)1 Old Tampa Bay	Alligator Creek	03100206000250	1.08	0.23
(b)1 Old Tampa Bay		03100206000083	1.08	0.23
(b)1 Old Tampa Bay		03100206000280	1.08	0.23
(b)2 Hillsborough Bay	Alafia River	03100204000004	1.62	1.28
(b)2 Hillsborough Bay		03100206000323	1.62	1.28
(b)2 Hillsborough Bay	BullFrog Creek	03100206000028	1.62	1.28
(b)2 Hillsborough Bay		03100206000044	1.62	1.28
(b)2 Hillsborough Bay	Hillsborough River	03100205000001	1.62	1.28
(b)3 Middle Tampa Bay	Little Manatee River	03100203000187	1.24	0.24
(b)3 Middle Tampa Bay		03100206000052	1.24	0.24
(b)3 Middle Tampa Bay	Wolf Branch	03100206000366	1.24	0.24
(b)3 Middle Tampa Bay	Piney Point Creek	03100206000501	1.24	0.24
(b)3 Middle Tampa Bay		03100206016249	1.24	0.24
(b)3 Middle Tampa Bay	Booker Creek	03100206001487	1.24	0.24
(b)7 Terra Ceia Bay	Frog Creek	03100206000467	1.10	0.14
(b)7 Terra Ceia Bay		03100206012310	1.10	0.14
(b)8 Manatee River Estuary	Williams Creek	03100202000895	1.80	0.37
(b)8 Manatee River Estuary	Manatee River	03100202000847	1.80	0.37

[a] Tributary names left blank are unnamed

2.14. Sarasota Bay

2.14.1. Proposed Numeric Nutrient Criteria

Newly-approved State water quality standards apply to Sarasota Bay (Subsection 62-302.532(c), F.A.C.).

2.14.2. Downstream Protective Values

Using geographic information systems, EPA integrated a waterway coverage with the estuary segments in Sarasota Bay and identified the pour point as the location where the waterway intersected with the estuary segment. In Sarasota Bay, EPA applied Florida's approved numeric criteria (Table 2-110) from the Florida downstream receiving segment as the proposed DPVs for TN and TP shown in Table 2-111.

Table 2-110. Newly-approved State water quality standards for Sarasota Bay

Estuary Segment	TN (mg/L)	TP (mg/L)
(c)1 Palma Sola Bay	0.93	0.26
(c)2 Sarasota Bay	*	0.19
(c)3 Roberts Bay	0.54	0.23
(c)4 Little Sarasota Bay	0.60	0.21
(c)5 Blackburn Bay	0.43	0.21

* Subsection 62-302.532(3)(i), F.A.C. applies

Table 2-111. Proposed DPVs for Sarasota Bay

Estuary Segment	Tributary[a]	USGS Reach Code	TN (mg/L)	TP (mg/L)
(c)2 Sarasota Bay		03100201002126	*	0.19
(c)2 Sarasota Bay		03100201001212	*	0.19
(c)2 Sarasota Bay		03100201009065	*	0.19
(c)2 Sarasota Bay		03100201009044	*	0.19
(c)3 Roberts Bay	Phillipe Creek	03100201004520	0.54	0.23
(c)5 Blackburn Bay	South Creek	03100201001251	0.43	0.21

[a] Tributary names left blank are unnamed
* TN criteria for this segment of Sarasota Bay varies according to mean color, region, and season; see 62-302.532(3)(i)

2.15. Charlotte Harbor/Estero Bay

2.15.1. Proposed Numeric Nutrient Criteria

Newly-approved State water quality standards apply to Charlotte Harbor/Estero Bay (Subsection 62-302.532(d), F.A.C.).

2.15.2. Downstream Protective Values

Using geographic information systems, EPA integrated a waterway coverage with the estuary segments in Charlotte Harbor/Estero Bay and identified the pour point as the location where the waterway intersected with the estuary segment. In Charlotte Harbor/Estero Bay, EPA applied Florida's approved numeric criteria (Table 2-112) from the Florida downstream receiving segment as the proposed DPVs for TN and TP. For pour points that drain to the Peace River, EPA applied the criteria from Florida's Charlotte Harbor Proper segment (which is the adjacent segment to the Peace River segment). For pour points in the Caloosahatchee River, EPA applied the criteria from Florida's San Carlos Bay segment (which is the adjacent segment to the

Caloosahatchee River segment). EPA's proposed DPVs for TN and TP are presented in Table 2-113.

Table 2-112. Newly-approved State water quality standards for Charlotte Harbor/Estero Bay

Estuary Segment	TN (mg/L)	TP (mg/L)
(d)1 Dona and Roberts Bay	0.42	0.18
(d)2 Upper Lemon Bay	0.56	0.26
(d)3 Lower Lemon Bay	0.62	0.17
(d)4 Charlotte Harbor Proper	0.67	0.19
(d)5 Pine Island Sound	0.57	0.06
(d)6 San Carlos Bay	0.56	0.07
(d)7 Tidal Myakka River	1.02	0.31
(d)8 Matlacha Pass	0.58	0.08
(d)9 Estero Bay (including tidal Imperial River)	0.63	0.07

Table 2-113. Proposed DPVs for Charlotte Harbor

Estuary Segment	Tributary/Structure[a]	USGS Reach Code	TN (mg/L)	TP (mg/L)
(d)1 Dona and Roberts Bay	Cow Pen Slough	03100201000018	0.42	0.18
(d)1 Dona and Roberts Bay	Curry Creek	03100201001281	0.42	0.18
(d)4 Charlotte Harbor Proper	Thornton Branch	03100101003360	0.67	0.19
(d)4 Charlotte Harbor Proper	Peace River	03100101003572	0.67	0.19
(d)4 Charlotte Harbor Proper	Shell Creek	03100101025442	0.67	0.19
(d)4 Charlotte Harbor Proper	Shell Creek	03100101025437	0.67	0.19
(d)4 Charlotte Harbor Proper	Myrtle Slough	03100101004051	0.67	0.19
(d)4 Charlotte Harbor Proper		03100101016221	0.67	0.19
(d)4 Charlotte Harbor Proper		03100103000637	0.67	0.19
(d)4 Charlotte Harbor Proper		03100101004013	0.67	0.19
(d)4 Charlotte Harbor Proper	Alligator Creek	03100103001794	0.67	0.19
(d)4 Charlotte Harbor Proper		03100201004302	0.67	0.19
(d)4 Charlotte Harbor Proper		03100103001484	0.67	0.19
(d)6 San Carlos Bay	Hickley Creek	03090205015983	0.56	0.07
(d)6 San Carlos Bay	Telegraph Creek	03090205015990	0.56	0.07
(d)6 San Carlos Bay	Trout River	03090205015999	0.56	0.07
(d)6 San Carlos Bay	Orange River	03090205017286	0.56	0.07
(d)6 San Carlos Bay	Manuel Branch	03090205010358	0.56	0.07
(d)6 San Carlos Bay	Stroud Creek	03090205010317	0.56	0.07
(d)6 San Carlos Bay	Hancock Creek	03090205010937	0.56	0.07
(d)6 San Carlos Bay	Whiskey Creek	03090205011704	0.56	0.07
(d)6 San Carlos Bay		03090205011367	0.56	0.07
(d)6 San Carlos Bay		03090205010938	0.56	0.07
(d)6 San Carlos Bay	Billy Creek	03090205010650	0.56	0.07
(d)6 San Carlos Bay	Cow Creek	03090204017420	0.56	0.07
(d)6 San Carlos Bay	Caloosahatchee River	03090205004078	0.56	0.07
(d)7 Tidal Myakka River	Myakka River	03100102000029	1.02	0.31
(d)7 Tidal Myakka River	Deer Prairie Creek	03100102001680	1.02	0.31
(d)7 Tidal Myakka River	Big Slough	03100102001297	1.02	0.31

Estuary Segment	Tributary/Structure[a]	USGS Reach Code	TN (mg/L)	TP (mg/L)
(d)7 Tidal Myakka River		03100103003331	1.02	0.31
(d)7 Tidal Myakka River	Rock Creek	03100102001139	1.02	0.31
(d)7 Tidal Myakka River	Sam Knight Creek	03100102001377	1.02	0.31
(d)8 Matlacha Pass		03100103006342	0.58	0.08
(d)8 Matlacha Pass		03100103006931	0.58	0.08
(d)9 Estero Bay (including tidal Imperial River)	Estero River	03090204024261	0.63	0.07

[a] Tributary names left blank are unnamed

In Lemon Bay, EPA applied Florida's approved numeric criteria (Table 2-112) from the Florida downstream receiving segment as the proposed DPVs for TN and TP (Table 2-114).

Table 2-114. Proposed DPVs for Lemon Bay

Estuary Segment	Tributary/Structure	USGS Reach Code	TN (mg/L)	TP (mg/L)
(d)2 Upper Lemon Bay	Forked Creek	03100201001292	0.56	0.26
(d)3 Lower Lemon Bay	Rock Creek	03100201001311	0.62	0.17
(d)3 Lower Lemon Bay	Coral Creek	03100201000001	0.62	0.17

2.16. Lake Worth Lagoon/Loxahatchee

2.16.1. Proposed Numeric Nutrient Criteria

Proposed numeric nutrient criteria for TN, TP, and chl-a in Lake Worth/Loxahatchee segments are summarized in Table 2-115.

Table 2-115. Proposed numeric nutrient criteria for Lake Worth/Loxahatchee segments

Segment Name	Segment Number	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
North Lake Worth Lagoon	1201	0.55	0.067	4.7
Central Lake Worth Lagoon	1202	0.57	0.089	5.3
South Lake Worth Lagoon	1203	0.48	0.034	3.6
Lower Loxahatchee	1301	0.68	0.028	2.7
Middle Loxahatchee	1302	0.98	0.044	3.9
Upper Loxahatchee	1303	1.25	0.072	3.6

2.16.2. General Characteristics

2.16.2.1. System Description

The Loxahatchee Estuary encompasses about 1.5 mi^2 (4 km^2 [400 ha]) and drains a watershed of about 270 mi^2 (700 km^2) in northeastern Palm Beach County and southeastern Martin County

(Howard et al. 2011). FDEP has designated several water bodies in the Loxahatchee Estuary system as OFWs.[69] The estuary is composed of North, Northwest, and Southwest Forks (SFWMD 2006).[70] The salinity in the North Fork is more uniform than the other forks because it receives less inflow (Noel et al. 1995). In 1985, part of the Northwest Fork [9.5 mi (15.3 km)] was designated as a National Wild and Scenic River, preserving much of the river corridor. Hydrologic modifications have diverted freshwater flows to the Southwest Fork and lowered groundwater levels, promoting saltwater intrusion during dry weather (Noel et al. 1995; SFWMD 2006). Ongoing remedial efforts attempt to increase the freshwater baseflow into the Northwest Fork (Howard et al. 2010). The construction of the C-18 Canal has diverted freshwater flow from the Northwest Fork into the Southwest Fork of the estuary through the S-46 structure (SFWMD 2006).

Lake Worth Lagoon is in the southeastern region of the Loxahatchee drainage in Palm Beach County, stretching 20 mi (32 km) along the eastern shore of south Florida. The lagoon is 0.5 mi (0.8 km) wide and has an average depth of 6 ft (1.8 m) (Braun 2006). Lake Worth Lagoon was historically a freshwater lake with no permanent surface connections to marine waters. The Lake Worth Inlet (also referred to as Palm Beach Inlet) was stabilized at its current location in 1918. South Lake Worth Inlet (also referred to as Boynton Inlet) was constructed in 1927 to improve water quality. Lake Worth Lagoon can be divided into three regions (north, central, and south).[71] Lake Worth Inlet is on the north portion of Lake Worth Lagoon and is the primary route for ocean water–freshwater exchange. Lake Worth Inlet also acts as the primary access to the Port of Palm Beach. The central portion is characterized by residential land use and an armored shoreline. The C-51 Canal is the major conduit for both freshwater and associated pollutants in this portion of the lagoon. The South Lake Worth Inlet is in the southernmost portion of the lagoon, in which the Boynton (C-16) Canal is the primary source of freshwater. WWTPs discharge directly and indirectly into Lake Worth Lagoon South (PBCDERM 2008).

From 1989 to 2004, the average annual rainfall in the Loxahatchee River watershed was 65 in (165 cm), two-thirds of which occurred between late May and the end of October. Average temperatures during the wet season are in the low 90s °F (low 30s °C) and in the low 80s °F (high 20s °C) during the dry season (FDEP and SFWMD 2010). Lake Worth Lagoon receives an average annual rainfall of 64.8 in (164.6 cm) (Dames & Moore and PBCDERM 1990).

Two aquifers are beneath the Loxahatchee River: the surficial and the Floridan aquifers. The surficial aquifer is shallow and is isolated from the Floridan aquifer by a thick clay boundary. The Floridan aquifer reaches a depth of 1,500 ft (460 m) and is composed of limestone (FDEP and SFWMD 2010). The Lake Worth Lagoon–Palm Beach Coast Basin contains three aquifer systems: Biscayne, surficial, and Floridan. The Biscayne aquifer is characterized by limestone, sand, and sandstone (FDEP 2006).

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[69] Section 62-302.700, F.A.C.

[70] The information presented in this system description was compiled to summarize local information pertaining to the proposed numeric nutrient criteria for this estuary. For more information on EPA's process of delineating, segmenting, and deriving numeric nutrient criteria for this estuary, please see Section 1.3.

[71] The information presented in this system description was compiled to summarize local information pertaining to the proposed numeric nutrient criteria for this estuary. For more information on EPA's process of delineating, segmenting, and deriving numeric nutrient criteria for this estuary, please see Section 1.3.

In the Loxahatchee watershed, approximately 25 percent of the land use is urban; the remainder is agricultural and other land uses (FDEP and SFWMD 2010). In the Lake Worth Lagoon basin, 47 percent of the land use is urban and 16 percent is agricultural (FDEP 2006). Lake Worth Lagoon is negatively affected by muck sediments, which cover large spans of shallow areas and dredge holes (PBCDERM 2008). In the Loxahatchee watershed, construction of canals and other water management activities such as ditching and draining have altered the water regime, and channelization has removed natural sandbars and oyster reefs to increase navigability. Such modifications have facilitated landward movement of seawater (FDEP and SFWMD 2010; VanArman et al. 2005). Lake Worth Lagoon was historically a freshwater lake, isolated from seawater exchange and receiving freshwater input primarily from surrounding wetlands. Human intervention altered the hydrology of the system by excavating two inlets to the ocean, converting Lake Worth to a saline lagoon (Crigger et al. 2005; PBCDERM 2008). Freshwater hydrology was altered by dredging canals and filling wetlands for development (PBCDERM 2008).

2.16.2.2. Impaired Waters[72]

Five Class III marine WBIDs in the combined Loxahatchee and Lake Worth Lagoon area have been listed for a nutrient-related parameter on Florida's CWA section 303(d) list approved by EPA. Of the five Class III marine WBIDs, two are impaired for DO (WBIDs 3224 and 3226E); one is impaired for nutrients and chl-a (WBID 3226); one is impaired for nutrients and DO (WBID 3226F); and one is impaired for nutrients, chl-a, and DO (WBID 3226D). No Class II WBIDs with nutrient-related impairments are documented for this area.[73]

No Class II or Class III marine WBIDs with nutrient-related TMDLs are documented in the region.[74]

2.16.2.3. Water Quality

In general, very little information was found in the available literature regarding observed water quality data for either Loxahatchee Estuary or Lake Worth Lagoon. Data from 2010 revealed that five of seven sampling zones in the Loxahatchee Estuary contained sample sites with DO levels below 5.0 mg/L (Howard et al. 2010). In Lake Worth Lagoon, data from 1994 to 2006 revealed an average DO concentration of 5.8 mg/L and a median concentration of 6.0 mg/L (PBCDERM 2008).

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[72] For more information about the data source, see Volume 1, Appendix A.

[73] The nutrient-related 303(d) list was a compilation of EPA-approved 303(d) listing information for nutrients, chl-a, and DO provided in three decision documents: September 2, 2009, Basin Groups 1, 2, and 5 EPA Decision Document (http://www.epa.gov/region4/water/tmdl/florida/documents/fl09303d_decisiondoc_090209.pdf);
May 13, 2010, Basin Group 3 EPA Decision Document (http://www.epa.gov/region4/water/tmdl/florida/documents/fl303d_%20partialapproval_decision_docs051410.pdf);
and the December 21, 2010, Basin Group 4 EPA Decision Document (http://www.epa.gov/region4/water/tmdl/florida/documents/group_4_final_dec_doc_and_partial_app_letter_12_21_10.pdf).

[74] TMDLs were identified in February 2011 by compiling nutrient-related draft/final TMDLs from the following three sources:
FDEP TMDL website (http://www.dep.state.fl.us/water/tmdl/index.htm)
EPA Region 4 website (http://www.epa.gov/region4/water/tmdl/florida/index.html)
EPA National WATERS expert query tool (http://www.epa.gov/waters/tmdl/expert_query.html).

Elevated chl-a was found throughout the Loxahatchee River watershed between October 2009 and September 2010. A preliminary review of chl-a over time suggests a trend of increasing concentrations relative to 1998–2002 levels. The 2010 annual geometric mean chl-a ranged from 2.7 µg/L in the marine zone to 11.0 µg/L in the brackish tributaries zone (Howard et al. 2010). In Lake Worth Lagoon, chl-a data from 2001 show mean and median concentrations of 4.4 and 3.2 µg/L, respectively (PBCDERM 2008). Annual chl-a levels were typically below 11 µg/L for estuarine systems and showed a decreasing trend between 2005 and 2008. Seasonally, chl-a concentrations peaked in August (RECOVER 2010).

In the Loxahatchee River watershed, the greatest TSS concentrations were found at the Jupiter Inlet and the central embayment (Noel et al. 1995). Percent light transmission was higher in 2010 than in earlier years, and TSS and turbidity values were lower in nearly all samples compared to earlier observations (Howard et al. 2010). In Lake Worth Lagoon, monitoring data showed an increasing trend in turbidity from 2001 to 2006. The mean and median turbidities were 4.6 and 3.1 NTU, respectively. Palm Beach County Department of Environmental Resources Management found that elevated turbidity was correlated with discharges from the C-51 Canal (PBCDERM 2008).

Elevated TN concentrations were observed at Loxahatchee River Environmental Control District sampling sites from 2004 to 2006 compared to 1998–2002 values. Those elevated nitrogen levels are attributed to hurricanes and excessive stormwater runoff that occurred in the same periods (FDEP and SFWMD 2010). Geometric mean TP values in 2010 ranged from 0.015 mg/L in the marine zone to 0.061 mg/L in the meso-/oligo-haline zone (Howard et al. 2010). No trends were observed for TN concentrations in Lake Worth Lagoon between 1999 and 2008. Seasonally, TN concentrations are slightly higher during the wet season (RECOVER 2010). The mean and median TP concentrations in Lake Worth Lagoon between 1994 and 2006 were 0.138 and 0.067 mg/L, respectively, with no apparent trend from 1994 to 2006 (PBCDERM 2008).

Changes in water patterns caused salinity to increase in the Northwest Fork of Loxahatchee Estuary (Dent 1997). Field data collected since 2002 from four sites along the Loxahatchee River measured mean salinity values of 31.8, 9.7, 1.5, and 0.5 PSU at river miles 1.77, 5.92, 8.13, and 9.12, respectively (Wan and Hu 2006). Conversely, altered hydrology has lowered the salinity of Lake Worth Lagoon (Crigger et al. 2005). Lake Worth Lagoon's salinity corresponds with seasonal variations in freshwater input (Dames & Moore and PBCDERM 1990; ECT and SEA 2008). During periods of reduced freshwater input, salinity levels are said to increase by 2 and 10 PSU. Three sections of the lagoon were found to have distinct salinity regimes: northern, central, and southern zones. In the northern zone, the salinity is generally 20–25 PSU, even during low salinity periods; in the central zone, salinity levels are typically around 25 PSU and drop to 5–15 PSU during low salinity periods; the southern zone experiences similar low salinity levels as the central zone, but salinity levels typically recover to 25–30 PSU. These cycles of variation, driven by weather and freshwater inflows, are said to occur approximately over a period of a few days to a month (ECT and SEA 2008).

2.16.2.4. Biological Characteristics

Coastal development removed mangrove and salt marsh habitats from all three forks of Loxahatchee Estuary. Development and bulkheading has nearly eliminated salt marsh and mangrove communities from the North and Southwest forks of the estuary (VanArman et al. 2005). Wetlands along the edges of Lake Worth Lagoon were filled for development; an action plan is in place to restore, create, and protect mangrove and *Spartina* spp. wetlands in Lake Worth Lagoon. Armoring of Lake Worth Lagoon shorelines increased from 79 percent to 87 percent between 1985 and 2001; mangrove acreage increased about 2 percent during that period (PBCDERM 2008).

All seven species of seagrasses (widgeon grass [*Ruppia maritime*], shoal grass [*Halodule wrightii*], manatee grass [*Syringodium filiforme*], turtle grass [*Thalassia testudinum*], paddle grass [*Halophila decipiens*], star grass [*H. engelmannii*], and Johnson's seagrass [*H. johnsonii*]) occur in the Loxahatchee Estuary; the dominant species is shoal grass (SFWMD 2006). Seagrass coverage in Lake Worth Lagoon is highest in the north with 65 percent of Lake Worth Lagoon's seagrasses; the remaining coverage is split between the central and southern segments, with 12 and 23 percent, respectively (FFWCC 2011). A 2007 survey found that seagrass coverage in Lake Worth Lagoon, totaling about 2.5 mi^2 (6.6 km^2 [1,626 ac]) in 2001, increased 0.07 mi^2 (0.17 km^2 [42 ac]) between 2001 and 2007 (Braun 2006; PBC 2008). A 2007 county mapping project identified 0.0066 mi^2 (0.0170 km^2 [4.2 ac]) of natural oyster reef in Lake Worth Lagoon (PBC 2008). A 1990 survey indicated that oyster reefs were present in the Northwest and Southwest forks, but are rare in the North Fork of Loxahatchee Estuary (SFWMD 2006).

Hanisak and Blair (1988) found 208 macroalgal taxa at the Lake Worth Inlet (excluding crustose corallines), 42 (20.2%), 19 (9.1%), and 147 (70.7%) belonging to the Chlorophyta, Phaeophyta, and Rhodophyta phyla, respectively. Most taxonomic diversity was found in late spring and summer months; the least diversity was found in late fall and winter (Hanisak and Blair 1988). Water column mean dissolved inorganic nitrogen (DIN) and soluble reactive phosphorus concentrations at the Princess Anne monitoring site (offshore of Riviera Beach, just north of Lake Worth Inlet) were greater than 0.014 mg/L as nitrogen and 0.003 mg/L as phosphorus (1.0 µM and 0.1 µM), respectively. Those levels are considered high for coral reefs and can sustain high growth of macroalgae (Lapointe 1997; Lapointe and Bedford 2010).

In 1965 there were over 250 fish species in the Loxahatchee River and Estuary (FDEP and SFWMD 2010). Lake Worth Lagoon is home to 195 fish species, and another 66 species exist in the vicinity of its inlets (Dames & Moore and PBCDERM 1990).

For a more detailed summary of this water body, see Volume 1, Appendix A.

2.16.3. Data Used

One data source specific to Lake Worth Lagoon/Loxahatchee was used in addition to those sources described in Section 1.4.3, as summarized in Table 2-116.

Table 2-116. Data source specific to Lake Worth Lagoon and Loxahatchee River models

Data	Source	Location Used
Measured Water Surface Elevation Data	South Florida Water Management District (SFWMD 2005)	Loxahatchee River Estuary model

2.16.4. Segmentation

The GIS isohaline analysis of Lake Worth lagoon yielded three segments with decreasing salinity from north to south. The north segment includes the Lake Worth Inlet. The segmentation scheme is consistent with the one proposed by FDEP (FDEP 2010a) for developing numeric nutrient criteria.

The GIS isohaline analysis of Loxahatchee Estuary yielded three segments with increasing salinity from mouth of the Loxahatchee River to the Jupiter Inlet, where it meets the Atlantic Ocean. The segmentation scheme is consistent with the one proposed by FDEP (FDEP 2010b), basing numeric nutrient criteria on trophic state variation in Loxahatchee Estuary. Figure 2-37 shows the resulting six segments for Lake Worth Lagoon/Loxahatchee Estuary.

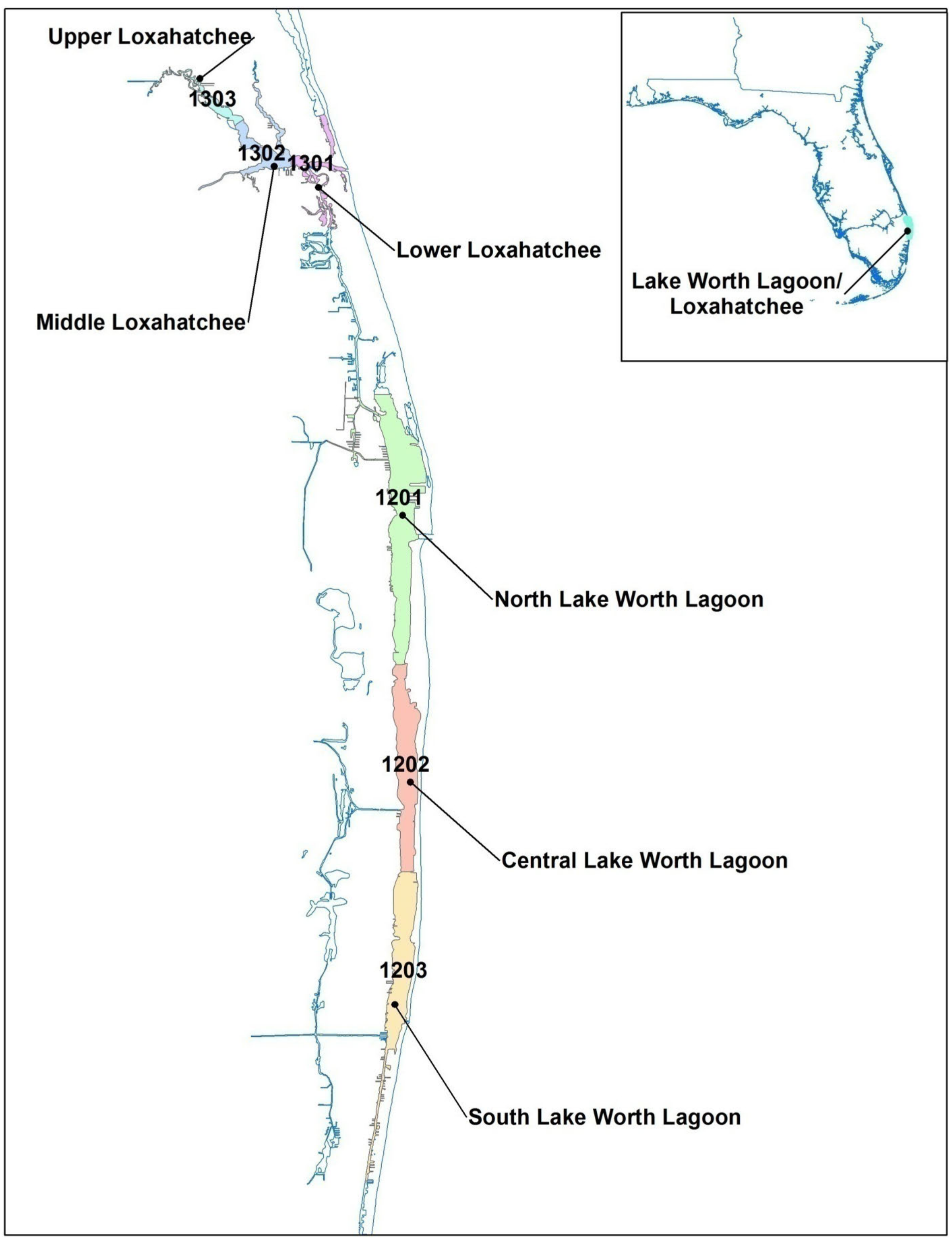

Figure 2-37. Results of Lake Worth Lagoon/Loxahatchee segmentation

2.16.5. Water Quality Targets

2.16.5.1. Seagrass Depth and Water Clarity Targets

Seagrass depth of colonization (measured as Z_c) and water clarity (measured as K_d) targets were established for the Lake Worth Lagoon/Loxahatchee Estuary segments by averaging the depth of colonization targets of WBIDs in each estuary segment, as shown in Table 2-117 (SJRWMD 2012).

Table 2-117. Lake Worth Lagoon/Loxahatchee seagrass depth and water clarity targets by segment

Estuary Segment	Depth of Colonization (Z_c) Target (m)	Light Attenuation Coefficient (K_d) Target (1/m)
1201	1.0	1.7
1202	0.8	2.1
1203	No target	-
1301	1.6	1.0
1302	1.25	1.3
1303	0.6	2.7

2.16.5.2. Chlorophyll *a* Target

To prevent nuisance algal blooms and protect the estuary's designated uses, chl-a levels must not exceed 20 µg/L more than 10 percent of the time. The rationale for that target is provided in Section 1.2.2.

2.16.5.3. Dissolved Oxygen Targets

On the basis of the rationale that sufficient DO is necessary to protect aquatic life, as described in Section 1.2.3, the following DO targets were established:

- Minimum allowable DO of 4.0 mg/L as a water column average for each estuary segment 90 percent of the time over the simulation's time span
- Daily average DO of 5.0 mg/L as a water column average for each estuary segment 90 percent of the time over the simulation's time span
- Minimum 3-hour average DO of 1.5 mg/L in the bottom two layers for each estuary segment over the simulation's time span

2.16.6. Results of Analyses

2.16.6.1. Mechanistic Model Analysis

Average load contributions to Lake Worth Lagoon from the Indian River watershed are shown in Table 2-118.

Table 2-118. Average load contributions to Lake Worth Lagoon from the Indian River watershed (2002–2009)

	TN Load[1] (kg/d; Mean ± se)			TP Load[1] (kg/d; Mean ± se)		
Year	Existing[2]	Background[3]	Anthropogenic[4]	Existing[2]	Background[3]	Anthropogenic[4]
2002	2,720 ± 168	1,157 ± 70	1,563 ± 102	177 ± 10	68 ± 4	109 ± 8
2003	3,197 ± 125	1,221 ± 46	1,976 ± 86	225 ± 6	68 ± 3	157 ± 6
2004	2,695 ± 228	946 ± 93	1,749 ± 137	180 ± 16	40 ± 3	140 ± 13
2005	3,203 ± 174	1,163 ± 69	2,039 ± 107	200 ± 12	54 ± 3	146 ± 9
2006	1,875 ± 105	566 ± 33	1,309 ± 76	101 ± 5	35 ± 2	66 ± 4
2007	2,294 ± 157	790 ± 59	1,504 ± 99	139 ± 10	35 ± 2	105 ± 8
2008	2,729 ± 204	936 ± 84	1,793 ± 122	159 ± 11	38 ± 3	121 ± 8
2009	1,721 ± 109	588 ± 39	1,133 ± 71	100 ± 6	29 ± 2	71 ± 4

[1] Annual average daily load from all terminal reaches
[2] Annual average daily load from all terminal reaches computed from existing model scenario
[3] Annual average daily load from all terminal reaches computed from background model scenario (remove anthropogenic sources/land use; use existing hydrology)
[4] Anthropogenic load computed as difference between existing and background load

Average load contributions to Loxahatchee River from the Indian River watershed are shown in Table 2-119.

Table 2-119. Average load contributions to Loxahatchee River from the Indian River watershed (2002–2009)

	TN Load[1] (kg/d; Mean ± se)			TP Load[1] (kg/d; Mean ± se)		
Year	Existing[2]	Background[3]	Anthropogenic[4]	Existing[2]	Background[3]	Anthropogenic[4]
2002	711 ± 39	290 ± 20	421 ± 19	36 ± 2	11 ± 1	25 ± 1
2003	674 ± 43	270 ± 22	404 ± 22	34 ± 2	11 ± 1	23 ± 1
2004	626 ± 66	256 ± 35	370 ± 31	32 ± 3	10 ± 1	22 ± 2
2005	938 ± 58	394 ± 29	543 ± 29	48 ± 3	16 ± 1	33 ± 2
2006	347 ± 15	118 ± 7	229 ± 8	17 ± 1	6 ± 0	11 ± 0
2007	621 ± 45	249 ± 21	372 ± 24	33 ± 2	11 ± 1	22 ± 1
2008	810 ± 59	328 ± 31	483 ± 29	40 ± 3	13 ± 1	27 ± 2
2009	500 ± 30	192 ± 14	308 ± 16	25 ± 1	8 ± 1	16 ± 1

[1] Annual average daily load from all terminal reaches
[2] Annual average daily load from all terminal reaches computed from existing model scenario
[3] Annual average daily load from all terminal reaches computed from background model scenario (remove anthropogenic sources/land use; use existing hydrology)
[4] Anthropogenic load computed as difference between existing and background load

For the purpose of developing numeric nutrient criteria, Lake Worth Lagoon and Loxahatchee were evaluated separately because they are two separate estuary models.

In Lake Worth Lagoon, DO targets were met for all segments on the basis of 2002–2009 nutrient loads. Light attenuation coefficient targets were met for segments 1201 and 1202 (a light attenuation coefficient target was not established for South Lake Worth Lagoon, segment 1203). The chl-a target was not met for segment 1202. Reduction runs were required to meet the chl-a target.

Table 2-120 identifies which targets were met under 2002–2009 nutrient loads and which targets were not met. Sensitivity analyses revealed the two available endpoints, chl-a and DO, were sensitive to changes in inputs of TN and TP.

Table 2-120. Water quality targets met for Lake Worth Lagoon based on mechanistic modeling

Segment	DO	Chl-a	K_d
1201	Yes	Yes	Yes
1202	Yes	No	Yes
1203	Yes	Yes	No target

In Loxahatchee River, chl-a and light attenuation coefficient targets were met for all segments on the basis of 2002–2009 nutrient loads. The DO targets were met on the basis of 2002–2009 nutrient loads, except for the daily water column average target of 5 mg/L, which could not be met for any segment using either the 2002–2009 nutrient loads scenario or the non-anthropogenic nutrient scenario. Table 2-121 identifies which targets were met under 2002–2009 nutrient loads and which targets were not met.

Sensitivity analyses revealed that in the Loxahatchee River DO was insensitive to changes in nutrients.

Table 2-121. Water quality targets met for Loxahatchee Estuary based on mechanistic modeling

Segment	DO	Chl-a	K_d
1301	No	Yes	Yes
1302	No	Yes	Yes
1303	No	Yes	Yes

A summary of candidate criteria for Lake Worth Lagoon segments is given in Table 2-122. A reduction scenario to meet the chl-a target was used to calculate candidate criteria for Lake Worth Lagoon.

Table 2-122. Summary of candidate criteria for Lake Worth Lagoon derived from mechanistic modeling

Segment	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
1201	0.55	0.067	4.7
1202	0.57	0.089	5.3
1203	0.48	0.034	3.6

Candidate criteria for Loxahatchee River segments are given in Table 2-123. 2002–2009 nutrient loads were used to calculate candidate criteria for Loxahatchee.

Table 2-123. Summary of candidate criteria for Loxahatchee derived from mechanistic modeling

Segment	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
1301	0.68	0.028	2.7
1302	0.98	0.044	3.9
1303	1.25	0.072	3.6

2.16.6.2. Statistical Model Analysis

Insufficient data were available to derive criteria in Lake Worth using statistical model analyses; however, analyses of data collected from the Loxahatchee River were possible. Annual geometric mean light attenuation coefficient increased with increased chl-a in all Loxahatchee segments (Figure 2-38). However, the slope of the light attenuation coefficient-chl-a relationship was not statistically significant in any of the segments, so no chl-a criteria associated with water clarity were derived.

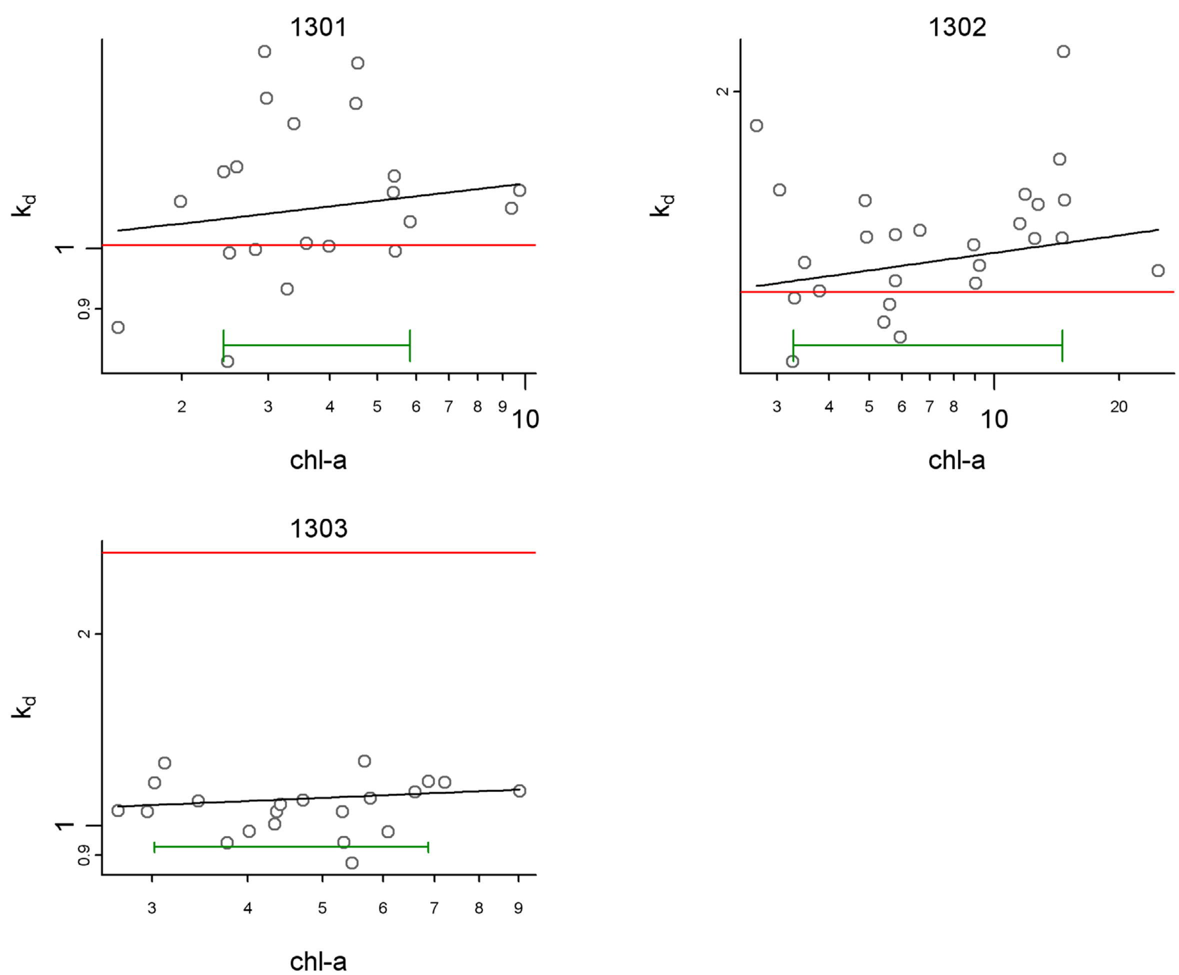

Figure 2-38. Relationships between annual corrected geometric K_d and chl-a in the Loxahatchee River. Solid black line: segment-wide relationship; red horizontal line: K_d target; red vertical arrow: chl-a concentrations associated with K_d target; green line segment: 5th to 95th percentile range of chl-a concentrations, open circles: observed annual geometric mean K_d, corrected for the effects of color and turbidity, and chl-a concentrations.

Estimates of chl-a concentrations associated with a phytoplankton bloom frequency of 10 percent were generally higher than the upper bound of the range of chl-a concentrations observed, except in segment 1302 (Figure 2-39). Hence, candidate chl-a criteria associated with the phytoplankton bloom endpoint were based on the upper bound of the data for all segments except for 1302. Based on these analyses, chl-a criteria were derived that protected both the water clarity and phytoplankton bloom endpoints (Table 2-124).

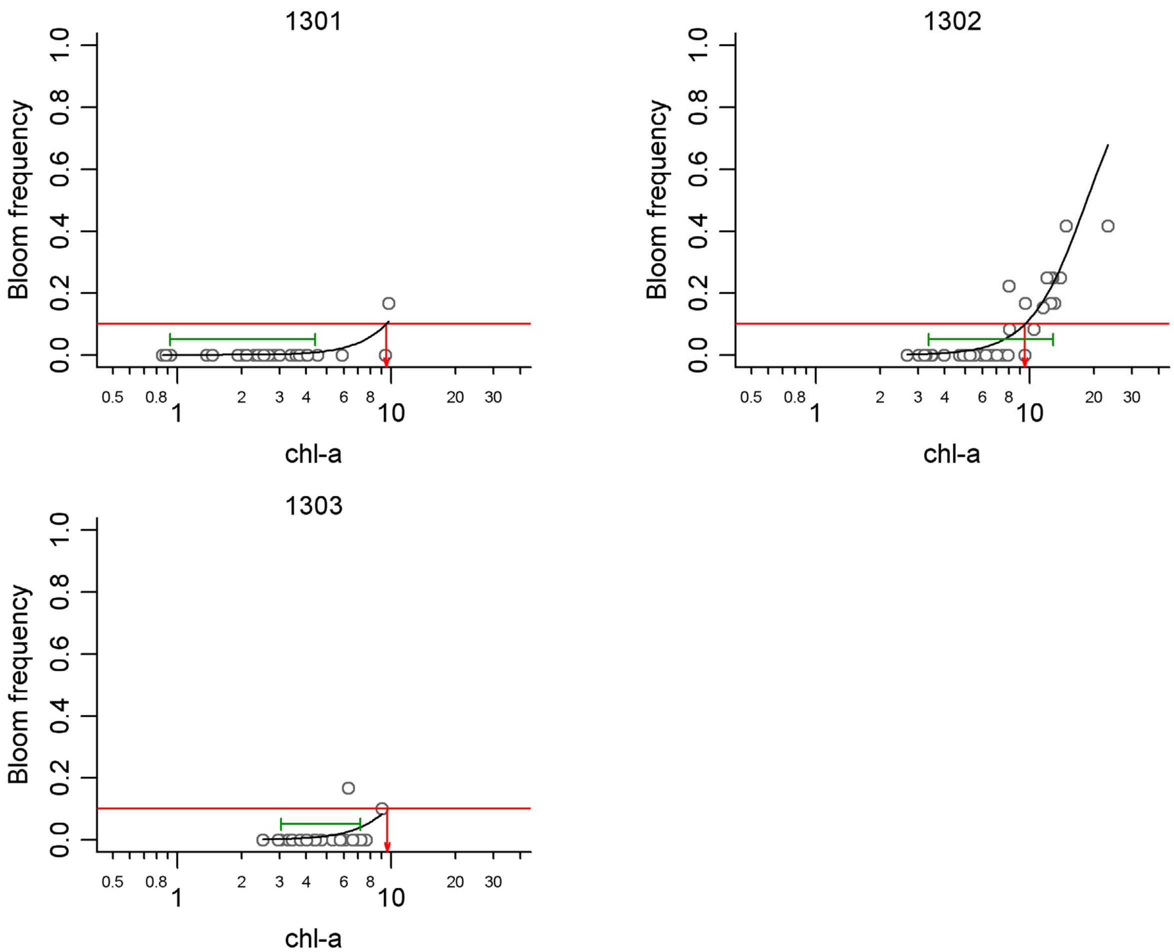

Figure 2-39. Estimates of annual geometric chl-a concentrations associated with bloom frequency of 0.1 in the Loxahatchee River. Red horizontal line: bloom frequency of 0.1, red vertical arrow: annual geometric mean chl-a concentration associated with 0.1 bloom frequency, green line segment: 5th to 95th percentile range of observed data.

Table 2-124. Summary of candidate chl-a criteria in the Loxahatchee River. Values with asterisks indicate that the predicted candidate criterion was greater than the upper bound of chl-a values, or less than the lower bound of chl-a values used in estimating the empirical relationship, so listed criterion is based on the upper or lower bound of the data.

Segment	Chl-a (clarity) (µg/L)	Chl-a (bloom) (µg/L)	Chl-a (final) (µg/L)
1301	-	4.4*	4.4
1302	-	9.5	9.5
1303	-	7.2*	7.2

In general, annual geometric mean chl-a increased with increasing concentrations of TN and TP in the Loxahatchee River (Figure 2-40). The relationship between TN and chl-a were not statistically significant, while relationships between TP and chl-a were statistically significant.

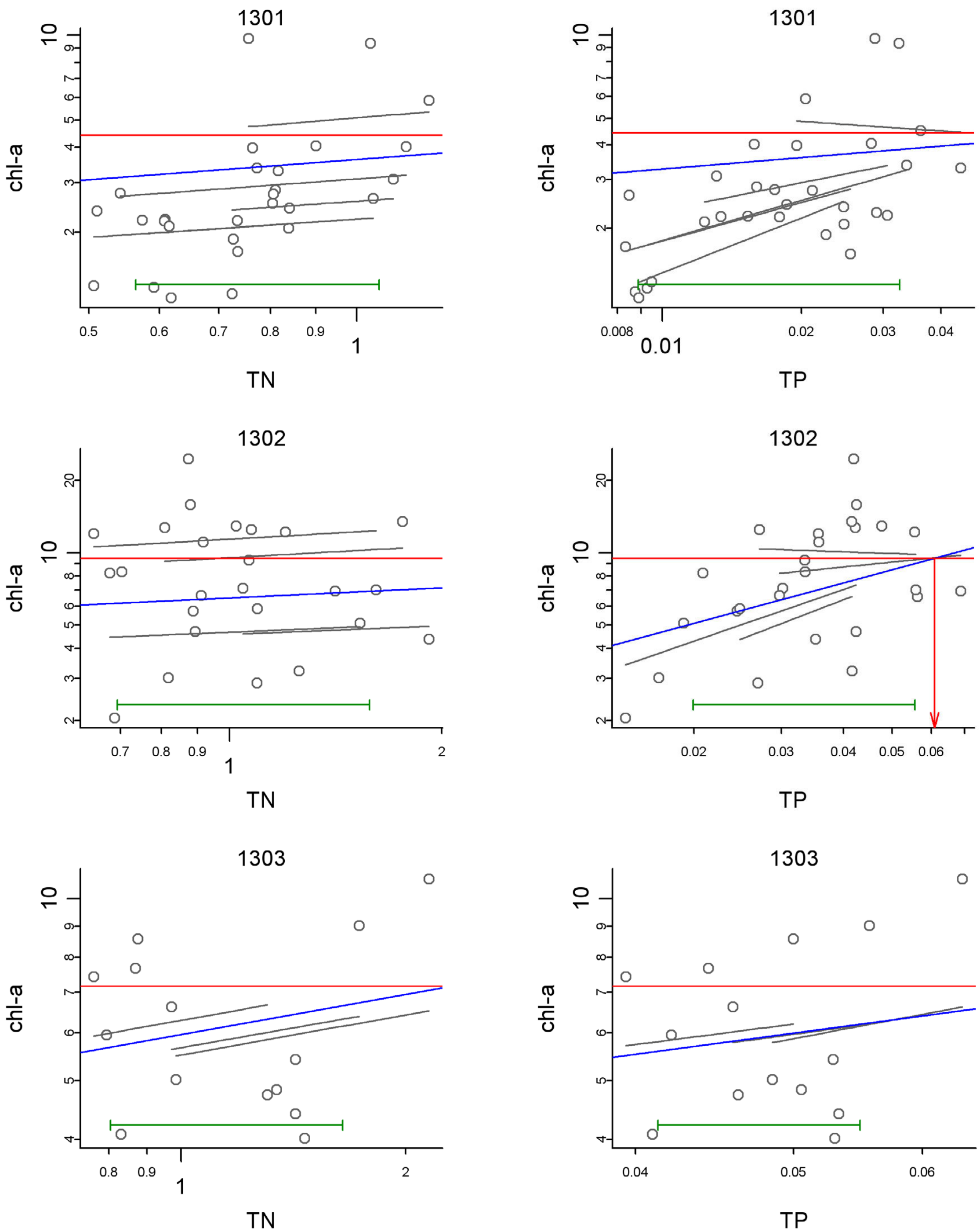

Figure 2-40. Relationships between TN, TP, and chl-a in the Loxahatchee River. Open circles: observed annual average values of TN, TP, and chl-a, red horizontal line: chl-a criterion, red vertical arrow: TN and TP criteria associated with chl-a criterion, blue line: estimated segment-wide relationship between TN, TP, and chl-a, grey lines: estimated station-specific relationships between TN, TP, and chl-a, green line segment: 5th to 95th percentile range of observed annual geometric mean TN and TP concentrations.

As with other estuaries, the variability among segments and among stations was expected given inherent differences in natural conditions among stations and segments. In segments 1301 and 1302, TP criteria associated with maintaining mean chl-a concentrations at the candidate criterion value were less than the lower bound of observed values. Hence, criteria values for TP in these segments were based on the lower bound of observed values. Conversely, in segment 1303, TP criteria associated with maintaining mean chl-a concentrations at the candidate criterion value were greater than the upper bound of observed values. Hence, criteria values in this segment are based on the upper bound of observed values. Proposed numeric nutrient criteria for TN and TP for the Loxahatchee River are summarized in Table 2-125.

Table 2-125. Summary of candidate TN and TP criteria in the Loxahatchee River. Values with asterisks indicate that the predicted candidate criterion was greater than the upper bound of observed values, or less than the lower bound of observed values used in estimating the empirical relationship, so listed criterion is based on the upper or lower bound of the data.

Segment	TN (mg/L)	TP (mg/L)
1301	-	0.033*
1302	-	0.056*
1303	-	0.055*

2.16.7. Application of Analyses for Proposed Numeric Nutrient Criteria

EPA evaluated various lines of evidence for deriving criteria in Lake Worth Lagoon. Data necessary to conduct statistical model analyses were not available for every segment. As a result, EPA derived the proposed numeric nutrient criteria for Lake Worth Lagoon using the mechanistic modeling results.

EPA evaluated three endpoints in the mechanistic modeling approach for Lake Worth Lagoon were data was present: (1) water clarity protective of historic depth of seagrasses, (2) chl-a concentrations associated with balanced phytoplankton biomass, and (3) DO concentrations sufficient to maintain aquatic life. EPA found that the chl-a target was not met under 2002–2009 loads, but it was sensitive to changes in nutrients. The water clarity and DO targets were achieved with the 2002–2009 nutrient loads, and were sensitive to changes in nutrients. A reduction in nutrients was applied to meet the chl-a target. The proposed criteria were derived to be protective of water clarity, chl-a, and DO concentrations. The values under mechanistic modeling below represent the 90th percentile annual geometric mean nutrient concentrations from the nutrient reduction scenario.

Proposed numeric nutrient criteria for TN, TP, and chl-a in Lake Worth Lagoon segments are summarized in Table 2-126.

Table 2-126. Proposed and candidate numeric nutrient criteria for Lake Worth Lagoon segments

		Proposed Criteria			Mechanistic Modeling		
SEGMENT	SEGMENT ID	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
North Lake Worth Lagoon	1201	0.55	0.067	4.7	0.55	0.067	4.7
Central Lake Worth Lagoon	1202	0.57	0.089	5.3	0.57	0.089	5.3
South Lake Worth Lagoon	1203	0.48	0.034	3.6	0.48	0.034	3.6

EPA evaluated various lines of evidence for deriving numeric nutrient criteria in the Loxahatchee River. Sufficient data were available to conduct statistical model analyses, but the relationship between nutrients and the biological endpoints applied were not statistically significant. As a result, EPA derived the proposed numeric nutrient criteria for Lake Worth Lagoon using the mechanistic modeling results.

EPA evaluated three endpoints in the mechanistic modeling approach for Loxahatchee: (1) water clarity protective of historic depth of seagrasses, (2) chl-a concentrations associated with balanced phytoplankton biomass, and (3) DO concentrations sufficient to maintain aquatic life. EPA found that water clarity and chl-a endpoints were met under 2002–2009 loads. EPA found that the DO endpoint was not met in all segments under the calibrated 2002–2009 nutrient loads and was insensitive to changes in nutrients loads. As a result, the DO endpoint was not used in Loxahatchee. The proposed criteria were derived to be protective of water clarity and chl-a concentrations. The values under the mechanistic modeling represent the 90th percentile annual geometric mean nutrient concentrations from the 2002–2009 modeled nutrient loads.

For the statistical modeling analyses, EPA evaluated chl-a concentrations that were protective of both phytoplankton bloom and water clarity endpoints, and found that the data did not support derivation of a chl-a criteria with respect to the water clarity endpoints. Instead, EPA derived a chl-a criterion to protect the bloom endpoint. TN was not significantly associated with chl-a concentrations in all segments, so no TN criteria were derived. TP concentrations derived to meet the chl-a criteria were greater than the upper bound of the data in two segments, so criteria were based on the upper bounds in those segments. In the other segment, TP concentrations supportive of the chl-a concentrations were less than the lower bound of observed values, so the lower bound of observed values was used.

Criteria derived from the mechanistic model were available in all segments and for TN and TP, so the proposed numeric nutrient criteria are based on the mechanistic model. Criteria derived from statistical models corroborated criteria derived using the mechanistic model (Table 2-127).

Table 2-127. Proposed and candidate numeric nutrient criteria for Loxahatchee River segments

		Proposed Criteria			Mechanistic Modeling			Statistical Modeling		
SEGMENT	SEGMENT ID	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
Lower Loxahatchee	1301	0.68	0.028	2.7	0.68	0.028	2.7	-	0.033	4.4
Middle Loxahatchee	1302	0.98	0.044	3.9	0.98	0.044	3.9	-	0.056	9.5
Upper Loxahatchee	1303	1.25	0.072	3.6	1.25	0.072	3.6	-	0.055	7.2

2.16.8. Downstream Protective Values

In Loxahatchee River/Lake Worth Lagoon mechanistic models were applied to derive the proposed DPVs for TN and TP shown in Table 2-128 and Table 2-129.

Table 2-128. Proposed DPVs for Lake Worth Lagoon

Tributary[a]	LSPC Model Watershed ID	USGS Reach Code	Estuary Segment	TN (mg/L)	TP (mg/L)
	190113	03090206028660	1203	1.12	0.0326
	190116	03090206036016	1201	0.83	0.0197
	190117	03090206036069	1202	1.05	0.0216
Boynton Beach Canal (c16)	c16	03090206028212	1203	0.29	0.0539
Earman River Canal (c17)	c17	03090206026649	1201	0.25	0.0287
West Palm Beach Canal (c51)	c51	03090206027480	1202	0.34	0.0510

[a] Tributary names left blank are unnamed

Table 2-129. Proposed DPVs for Loxahatchee River

Tributary[a]	LSPC Model Watershed ID	USGS Reach Code	Estuary Segment	TN (mg/L)	TP (mg/L)
	190076	03090206033866	1301	0.81	0.023
	190078	03090206035996	1301	1.17	0.029
North Fork Loxahatchee River	190080	03090206033910	1302	1.40	0.028
	190086	03090206033915	1302	1.31	0.028
	190088	03090206033871	1303	1.03	0.114
	190089	03090206023926	1303	1.22	0.169
	190093	03090206033909	1302	0.80	0.032
	190099	03090206033922	1302	1.45	0.030
C-46	c46	03090206024980	1302	1.07	0.036

[a] Tributary names left blank are unnamed

2.16.9. References

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2.17. St. Lucie Estuary

2.17.1. Proposed Numeric Nutrient Criteria

Proposed numeric nutrient criteria for TN, TP, and chl-a in St. Lucie segments are summarized in Table 2-130.

Table 2-130. Proposed numeric nutrient criteria for St. Lucie segments

Segment Name	Segment Number	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
Lower St. Lucie	1401	0.58	0.045	5.3
Middle St. Lucie	1402	0.90	0.120	8.4
Upper St. Lucie	1403	1.22	0.197	8.9

2.17.2. General Characteristics

2.17.2.1. System Description

Located in Martin and St. Lucie counties in southeast Florida, St. Lucie Estuary is a major tributary of the southern Indian River Lagoon (SJRWMD 2010). The St. Lucie Estuary is north of Lake Worth and east of Lake Okeechobee, from which it receives lake water via the C-44 Canal. The inland portion of the estuary is composed of two forks—North and South—which converge at the Roosevelt Bridge to form a single water body that connects the St. Lucie Estuary to the Indian River Lagoon (Parmer et al. 2008).[75] Ocean access and tidal exchange between the St. Lucie Estuary and the Atlantic Ocean occur through two man-made waterways: St. Lucie and Fort Pierce Inlets (FDEP 2011). Five major rivers, creeks, and canals provide drainage for the watershed: Ten Mile Creek, C-24 Canal, C-23 Canal, Old South Fork, and the St. Lucie Canal (C-44) (FDEP 2010). The total surface water area of the St. Lucie Estuary is approximately 18 mi^2 (29 km^2), and the mean water depth is 7.9 ft (2.4 m) (FDEP 2010; Ji et al. 2007). The St. Lucie River watershed encompasses approximately 1,050 mi^2 (2,720 km^2) (FDEP 2004, 2011).

St. Lucie County receives an annual average rainfall of 49.8 in (126.4 cm), with the highest monthly averages occurring during summer and fall (Visit St. Lucie 2012). The average annual temperature in St. Lucie County is 73.7 °F (23.2 °C) with a mean annual maximum of 81.6 °F (27.6 °C) and mean annual minimum of 65.8 °F (18.8 °C) (Visit St. Lucie 2012). In 2004 the largest land uses were citrus (22.6%; 182 mi^2 [471 km^2 (116,442 ac)]), improved pasture (20.7%; 166 mi^2 [430 km^2 (106,321 ac)]), urban (16.3%; 131 mi^2 [339 km^2 (83,861 ac)]), and wetland natural areas (11.9%; 95 mi^2 [247 km^2 (61,052 ac)]) (SFWMD et al. 2009). The urban areas are primarily distributed throughout the eastern region of the watershed (FDEP 2010). For planning and management purposes, FDEP divided the St. Lucie watershed into six individual planning units on the basis of their dominant land uses (Parmer et al. 2008). The planning units and associated primary land use are North St. Lucie (urban and agriculture: 35% each), South

[75] The information presented in this system description was compiled to summarize local information pertaining to the proposed numeric nutrient criteria for this estuary. For more information on EPA's process of delineating, segmenting, and deriving numeric nutrient criteria for this estuary, please see Section 1.3.

St. Lucie (agriculture: 32%; urban: 26%), C-23 (agriculture: 64%), C-24 (agriculture: 61%), C-44 (agriculture: 63%), and Coastal (urban: 26%) (FDEP 2004).

Key processes affecting the hydrodynamic transport and contaminant dilution in the estuary are estuarine stratification, lateral inflows, flushing time, and salinity intrusion (Ji et al. 2007). In 1999 and 2000 lateral inflows contributed 23 and 37 percent of total freshwater discharge into the St. Lucie Estuary, respectively. Lateral inflows can significantly decrease salinity levels and affect water elevation; during flooding periods, surface elevations can rise to 4 in (10 cm) (Ji et al. 2007). The St. Lucie Estuary has a calculated mean flushing time of 3.2–47 days, on the basis of calculations by different formulas (Ji et al. 2007). The St. Lucie Estuary is also characterized by semi-diurnal tides with a tidal range of approximately 1.8 ft (0.5 m) (FDEP 2010).

Groundwater is the primary source of drinking water in the St. Lucie Basin. Two major aquifer systems in the basin are the surficial and the Floridan aquifer systems. Floridan aquifer groundwater is characterized by high dissolved solids and chloride concentrations, so its use as a drinking water supply is limited. St. Lucie and Martin counties in their entirety have been designated Critical Water Supply Problem Areas. Saltwater intrusion is the primary threat to increased water use demands along the coast (FDEP 2004).

2.17.2.2. Impaired Waters[76]

Two Class III marine WBIDs in the St. Lucie Estuary have been listed for a nutrient-related parameter on Florida's CWA section 303(d) list approved by EPA. Of the two Class III marine WBIDs, one is impaired for DO (WBID 3208A) and one is impaired for nutrients and chl-a (WBID 3208). No Class II WBIDs with nutrient-related impairments are documented for this area.[77]

One nutrient-related TMDL for Class II or Class III marine WBIDs exists in the St. Lucie Estuary watershed, the final *St. Lucie Basin Nutrient and DO TMDL,* covering six Class III marine WBIDs (3193, 3194, 3194B, 3210, 3210A, 3211).[78]

2.17.2.3. Water Quality

Water quality in the St. Lucie Estuary is affected by both point sources (e.g., permitted discharges) and nonpoint sources (e.g., stormwater runoff, leaking septic tanks) of pollution (IRLNEP and SJRWMD 2009). The estuary has experienced nutrient pollution over many years because of urbanization and agricultural activities in the watershed (Parmer et al. 2008). Signs of

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[76] For more information about the data source, see Volume 1 Appendix A.

[77] The nutrient-related 303(d) list was a compilation of EPA-approved 303(d) listing information for nutrients, chl-a, and DO provided in three decision documents: September 2, 2009, Basin Groups 1, 2, and 5 EPA Decision Document (http://www.epa.gov/region4/water/tmdl/florida/documents/fl09303d_decisiondoc_090209.pdf);
May 13, 2010, Basin Group 3 EPA Decision Document
(http://www.epa.gov/region4/water/tmdl/florida/documents/fl303d_%20partialapproval_decision_docs051410.pdf);
and the December 21, 2010, Basin Group 4 EPA Decision Document
(http://www.epa.gov/region4/water/tmdl/florida/documents/group_4_final_dec_doc_and_partial_app_letter_12_21_10.pdf).

[78] TMDLs were identified in February 2011 by compiling nutrient-related draft/final TMDLs from the following three sources:
FDEP TMDL website (http://www.dep.state.fl.us/water/tmdl/index.htm)
EPA Region 4 website (http://www.epa.gov/region4/water/tmdl/florida/index.html)
EPA National WATERS expert query tool (http://www.epa.gov/waters/tmdl/expert_query.html).

water quality degradation include the decline of seagrasses and oysters, algal blooms, fish kills, and low diversity of benthic macroinvertebrates (Graves et al. 2002).

For most water quality parameters in the St. Lucie Estuary, there is a spatial gradient from the North and South Forks to the Outer Estuary (FDEP 2010). Inner regions of the estuary that are close to freshwater inflows tend to exhibit higher nutrient loads and reduced water quality compared to Outer Estuary regions and the Indian River Lagoon (Tunberg et al. 2009). Between 1997 and 2009, median concentrations of chl-a, TN, total Kjeldhal nitrogen (TKN), and nitrate+nitrite were generally higher in the North and South Forks and declined in the Middle and Outer Estuary regions, where salinity was higher (FDEP 2010).

In 1996 approximately 10 to 25 percent of the St. Lucie River was anoxic (DO = 0 mg/L) and 25 to 50 percent was hypoxic (DO ≤ 2 mg/L) (NOAA 1996). During a 2005–2009 study period, DO measurements in the St. Lucie Estuary reflected seasonal fluctuations. During the wet season, DO levels generally decreased (Tunberg et al. 2009).

Data from 1997 to 2009 showed that chl-a ranges and the median concentration were the largest in the North Fork (12.9 µg/L), followed by the South Fork (median 9.8 µg/L), and Middle Estuary (median 7 µg/L). The Outer Estuary showed the lowest range and median concentration (3.3 µg/L). In the North Fork, South Fork, and Middle Estuary regions of the St. Lucie Estuary, chl-a concentrations occasionally exceed 10 µg/L, sometimes reaching as high as 60 µg/L (FDEP 2010).

Water clarity is generally reduced during periods of high freshwater discharge (FDEP 2010). In 1996, NOAA reported that turbidity levels in the St. Lucie River were high at Secchi depths ≤ 3.3 ft (1 m) (NOAA 1996). A comprehensive survey on seagrass cover in the Indian River Lagoon reported that there has been a decline in TSS between 1992 and 2008, potentially attributed to drought conditions between 2005 and 2007 (FFWCC 2011).

Total dissolved nitrogen (TDN) concentrations in the St. Lucie River in 1996 were considered high at ≥ 1 mg/L (NOAA 1996). TN concentrations in the waters draining from canals C-23, C-24, and C-44 often exceed 2.5 mg/L (Graves and Strom 1992).

In 1996, total dissolved phosphorus (TDP) concentrations in the St. Lucie River were considered high at levels ≥ 0.1 mg/L (NOAA 1996). Total phosphate concentrations in the waters draining from C-23, C-24, and C-44 canals often exceed 0.25 mg/L, contributing at least 217,000 lb (98,400 kg) of phosphorus to St. Lucie Estuary annually (Graves and Strom 1992). As of 2008, the median TP concentration at the Roosevelt Bridge and at the confluence between the St. Lucie Estuary and Indian River Lagoon were 164 µg/L and 84 µg/L, respectively (Parmer et al. 2008).

During a 2005 to 2009 study of the St. Lucie Estuary, seasonal salinity fluctuations in St. Lucie Estuary were primarily driven by freshwater inflow (Tunberg et al. 2009). From 1997 to 2009 the St. Lucie Estuary had a range of salinities from less than 1 PSU to more than 28 PSU. The median salinity of the North Fork (2.7 PSU) was more than double the median of the South Fork (1.1 PSU) (FDEP 2010).

2.17.2.4. Biological Characteristics

The St. Lucie Estuary and Indian River Lagoon collectively form the most biologically diverse estuary in North America (IRLNEP and SJRWMD 2009; SJRWMD 2010). The North Fork St. Lucie River is designated a state aquatic reserve and is part of Florida's Save Our Rivers program (SFWMD 2009). FDEP highlighted the following three major watershed factors that affect the ecological health of the St. Lucie Estuary: (1) excessive nutrient loadings, (2) freshwater discharges into the basin, and (3) undesirable low flows to the St. Lucie Estuary (FDEP 2010).

Brackish areas in the watershed support extensive mangrove swamps (Boning 2007). However, shoreline habitat created by mangrove wetlands and bank vegetation has decreased along the tributaries of the St. Lucie Estuary and Indian River Lagoon (Sime 2005). In 2007 sparse patches of SAV, with less than 10 percent coverage, were present in the Lower and Middle estuary. No SAV was found in either the North or South forks. The dominant species was Johnson's seagrass (*Halophila johnsonii*) (RECOVER 2010). FFWCC reported that the overall seagrass coverage in the southern Indian River Lagoon increased by 13.3 percent per year between 2005 and 2007. The most common seagrass species found in the St. Lucie Inlet were shoal grass (*Halodule wrightii*) and manatee grass (*Syringodium filiforme*) (FFWCC 2011).

NOAA's 1996 *Estuarine Eutrophication Survey* identified the occurrence of nuisance and toxic algae in the St. Lucie River (NOAA 1996). Water samples revealed *Cryptoperidiniopsis* in Florida's Indian and St. Lucie rivers (Anonymous 1998). Millie et al. (2004) concluded that water quality and phytoplankton dynamics were related to seasonal hydrologic and salinity conditions between March 2000 and March 2001. The phytoplankton community was dominated by diatoms and dinoflagellates. A few select taxa demonstrated distinct wet/dry seasonality in phytoplankton biomass (Millie et al. 2004). Increases in available nitrogen loads to the ecosystem likely enhance algal bloom potential (Lin et al. 2008). Between 1997 and 2003, more than 0.14 mi^2 (0.36 km^2 [90 ac]) of live oyster reefs were lost because of low salinities (1–5 PSU) (FDEP 2010). Salinity between 10 and 30 PSU is optimal for eastern oysters to thrive (Parker and Geiger 2009).

Benthic macroinvertebrate communities in the St. Lucie Estuary and Indian River Lagoon reportedly are affected by nutrient loads, salinity patterns, and DO levels associated with freshwater inflows. Freshwater inflows are highly seasonal, so they create seasonal fluctuations in the number of taxa and abundance of benthic populations. Warmer, wetter months showed decreased benthic species richness and abundance in the St. Lucie Estuary during a 2005 to 2009 study. In general, sites in the North Fork, South Fork, and Middle Estuary demonstrated the lowest species richness throughout the St. Lucie Estuary and Indian River Lagoon systems (Tunberg et al. 2009).

More than 500 fish species have been reported within a 2-mile (3.2-km) radius of the confluence of the St. Lucie Estuary and the Indian River Lagoon (SFWMD 2009). Commonly occurring fish species in the St. Lucie River include brown bullhead, bowfin, Florida gar, golden shiner, sailfin molly, and white catfish (Boning 2007). FFWCC Research Institute's Fisheries Independent Monitoring Program was established in April 1998 to monitor trends in estuarine fish abundance over time. From 2001 to 2008, between 86 and 109 species were identified in the St. Lucie Estuary (FDEP 2010). The number of lesioned fish is positively correlated with volume of freshwater influx into the St. Lucie Estuary and could be associated with the pathogen *Aphanomyces invadans* (Sime 2005; Sosa et al. 2007).

For a more detailed summary of this water body, see Volume 1, Appendix A.

2.17.3. Data Used

No additional data sources specific to the St. Lucie estuary are available; see the general data sources described in Section 1.4.3.

2.17.4. Segmentation

The GIS isohaline analysis of St. Lucie Estuary yielded three segments with increasing salinity from the mouth of the St. Lucie River to St. Lucie Inlet where it meets the Atlantic Ocean. Figure 2-41 shows the three segments for St. Lucie Estuary.

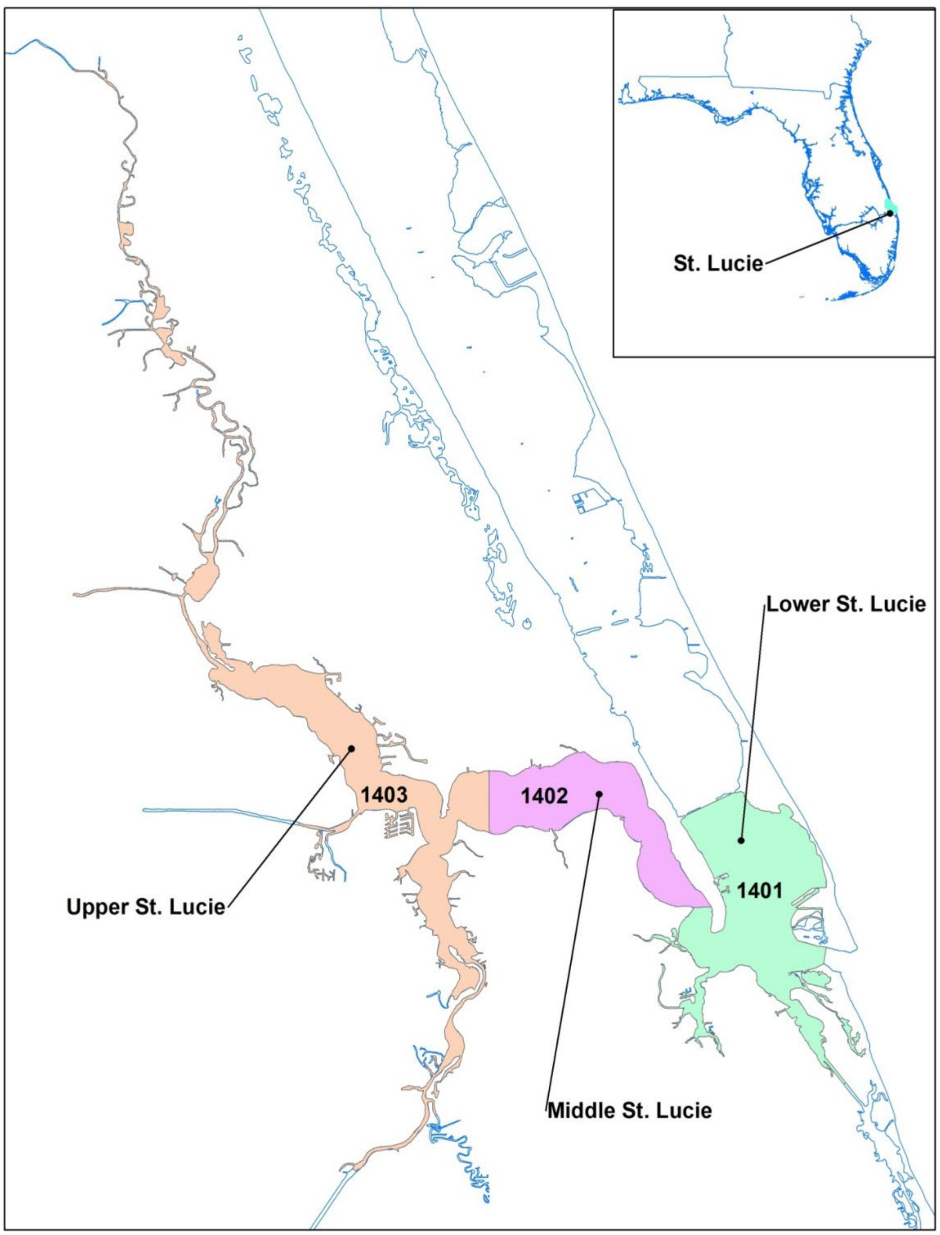

Figure 2-41. Results of St. Lucie Estuary segmentation

2.17.5. Water Quality Targets

2.17.5.1. Seagrass Depth and Water Clarity Targets

Seagrass depth of colonization (measured as Z_c) and water clarity (measured as K_d) targets were established for the St. Lucie Estuary segments by averaging the depth of colonization targets of WBIDs in each estuary segment, as shown in Table 2-131 (SJRWMD 2012).

Table 2-131. St. Lucie seagrass depth and water clarity targets by segment

Estuary Segment	Depth of Colonization (Z_c) Target (m)	Light Attenuation Coefficient (K_d) Target (1/m)
1401	2.06	0.8
1402	No target	-
1403	No target	-

2.17.5.2. Chlorophyll a Target

To prevent nuisance algal blooms and protect the estuary's designated uses, chl-a levels must not exceed 20 µg/L more than 10 percent of the time. The rationale for that target is provided in Section 1.2.2.

2.17.5.3. Dissolved Oxygen Targets

On the basis of the rationale that sufficient DO is necessary to protect aquatic life, as described in Section 1.2.3, the following DO targets were established:

- Minimum allowable DO of 4.0 mg/L as a water column average for each estuary segment 90 percent of the time over the simulation's time span
- Daily average DO of 5.0 mg/L as a water column average for each estuary segment 90 percent of the time over the simulation's time span
- Minimum 3-hour average DO of 1.5 mg/L in the bottom two layers for each estuary segment over the simulation's time span

2.17.6. Results of Analyses

2.17.6.1. Mechanistic Model Analysis

Average load contributions to St. Lucie Estuary from the Indian River watershed are shown in Table 2-132.

Table 2-132. Average load contributions from the Indian River watershed (2002–2009)

Year	TN Load[1] (kg/d; Mean ± se)			TP Load[1] (kg/d; Mean ± se)		
	Existing[2]	Background[3]	Anthropogenic[4]	Existing[2]	Background[3]	Anthropogenic[4]
2002	3,840 ± 212	2,009 ± 124	1,830 ± 100	423 ± 25	222 ± 16	201 ± 11
2003	6,300 ± 301	4,400 ± 221	1,901 ± 111	576 ± 35	371 ± 24	205 ± 12
2004	7,324 ± 550	5,426 ± 447	1,899 ± 158	575 ± 67	345 ± 44	229 ± 24
2005	10,508 ± 406	7,935 ± 321	2,573 ± 136	867 ± 51	568 ± 35	299 ± 18
2006	1,790 ± 104	1,027 ± 84	763 ± 33	152 ± 11	90 ± 7	62 ± 4
2007	1,854 ± 143	805 ± 71	1,049 ± 74	174 ± 14	71 ± 6	103 ± 8
2008	4,034 ± 402	2,145 ± 231	1,889 ± 182	379 ± 43	187 ± 24	193 ± 20
2009	1,998 ± 120	896 ± 65	1,103 ± 61	227 ± 15	111 ± 9	116 ± 7

[1] Annual average daily load from all terminal reaches
[2] Annual average daily load from all terminal reaches computed from existing model scenario
[3] Annual average daily load from all terminal reaches computed from background model scenario (remove anthropogenic sources/land use; use existing hydrology)
[4] Anthropogenic load computed as difference between existing and background load

In the St. Lucie Estuary, DO, chl-a, and light attenuation coefficient targets were met for all segments using 2002–2009 nutrient loads. Table 2-133 identifies which targets were met under 2002–2009 nutrient loads and which targets were not met.

Table 2-133. Water quality targets met for the St. Lucie Estuary based on mechanistic modeling

Segment	DO	Chl-a	K_d
1401	Yes	Yes	Yes
1402	Yes	Yes	No target
1403	Yes	Yes	No target

Candidate criteria for St. Lucie Estuary segments based on 2002–2009 nutrient loads are given in Table 2-134.

Table 2-134. Summary of candidate criteria for the St. Lucie Estuary derived from mechanistic modeling

Segment	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
1401	0.58	0.045	5.3
1402	0.90	0.120	8.4
1403	1.22	0.197	8.9

2.17.6.2. Statistical Model Analysis

Sufficient data were available to derive criteria for St. Lucie Estuary using statistical models. In segment 1401, where a water clarity target was available, the light attenuation coefficient increased with increasing chl-a, but this relationship was not statistically significant (Figure 2-42). Hence, no chl-a candidate criterion associated with water clarity was derived.

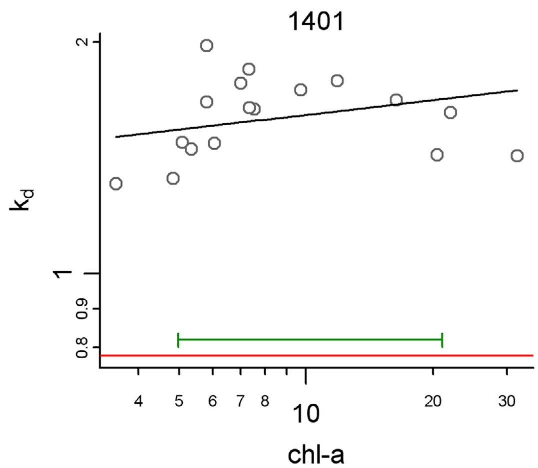

Figure 2-42. Relationships between corrected annual geometric K_d and chl-a in segment 1401, St. Lucie Estuary. Solid black line: segment-wide relationship; red horizontal line: K_d target; red vertical arrow: chl-a concentrations associated with K_d target; green line segment: 5th to 95th percentile range of chl-a concentrations, open circles: observed annual geometric mean K_d, corrected for the effects of color and turbidity, and chl-a concentrations.

The frequency of phytoplankton blooms was strongly associated with annual geometric mean chl-a concentrations in all segments of St. Lucie, and chl-a criteria were derived such that average bloom frequencies were 10 percent in segments 1401 and 1403 (Figure 2-43). In segment 1402, the derived candidate chl-a criterion was greater than the upper bound of observed chl-a concentrations and therefore, the candidate chl-a criterion for this segment was set the upper bound of observed chl-a concentrations.

Final candidate chl-a concentrations were derived for all segments that were protective of both the bloom and water clarity endpoints, where applicable (Table 2-135).

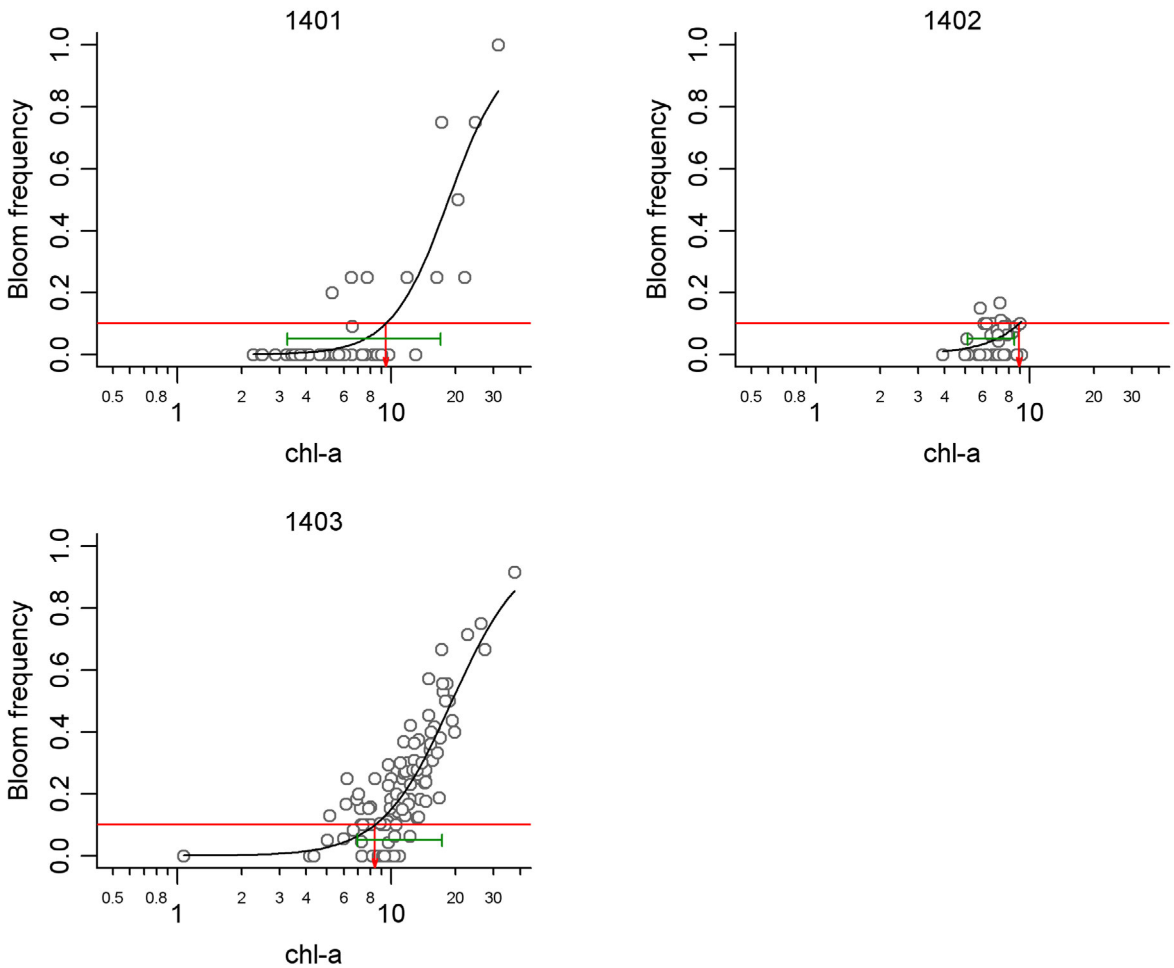

Figure 2-43. Estimates of annual geometric chl-a concentrations associated with bloom frequency of 0.1 in St. Lucie. Red horizontal line: bloom frequency of 0.1, red vertical arrow: annual geometric mean chl-a concentration associated with 0.1 bloom frequency, green line segment: 5th to 95th percentile range of observed data.

Table 2-135. Summary of candidate chl-a criteria in St. Lucie Estuary. No water clarity targets were available in segments 1402 and 1403, so no chl-a criteria were calculated based on clarity. Values with asterisks indicate that the predicted candidate criterion was greater than the upper bound of chl-a values, or less than the lower bound of chl-a values used in estimating the empirical relationship, so listed criterion is based on the upper or lower bound of the data.

Segment	Chl-a (clarity) (µg/L)	Chl-a (bloom) (µg/L)	Chl-a (final) (µg/L)
1401	-	9.4	9.4
1402	-	8.5*	8.5
1403	-	8.4	8.4

Increased concentrations of TN and TP were generally associated with increased chl-a concentrations in all segments in St. Lucie, but relationships between chl-a and TN were not statistically significant in segment 1401 and 1402 (Figure 2-44). In segments 1401 and 1402, derived criteria were greater than the upper bound of observed values and in segment 1403 derived criteria were less than the lower bound of observed values (Table 2-136).

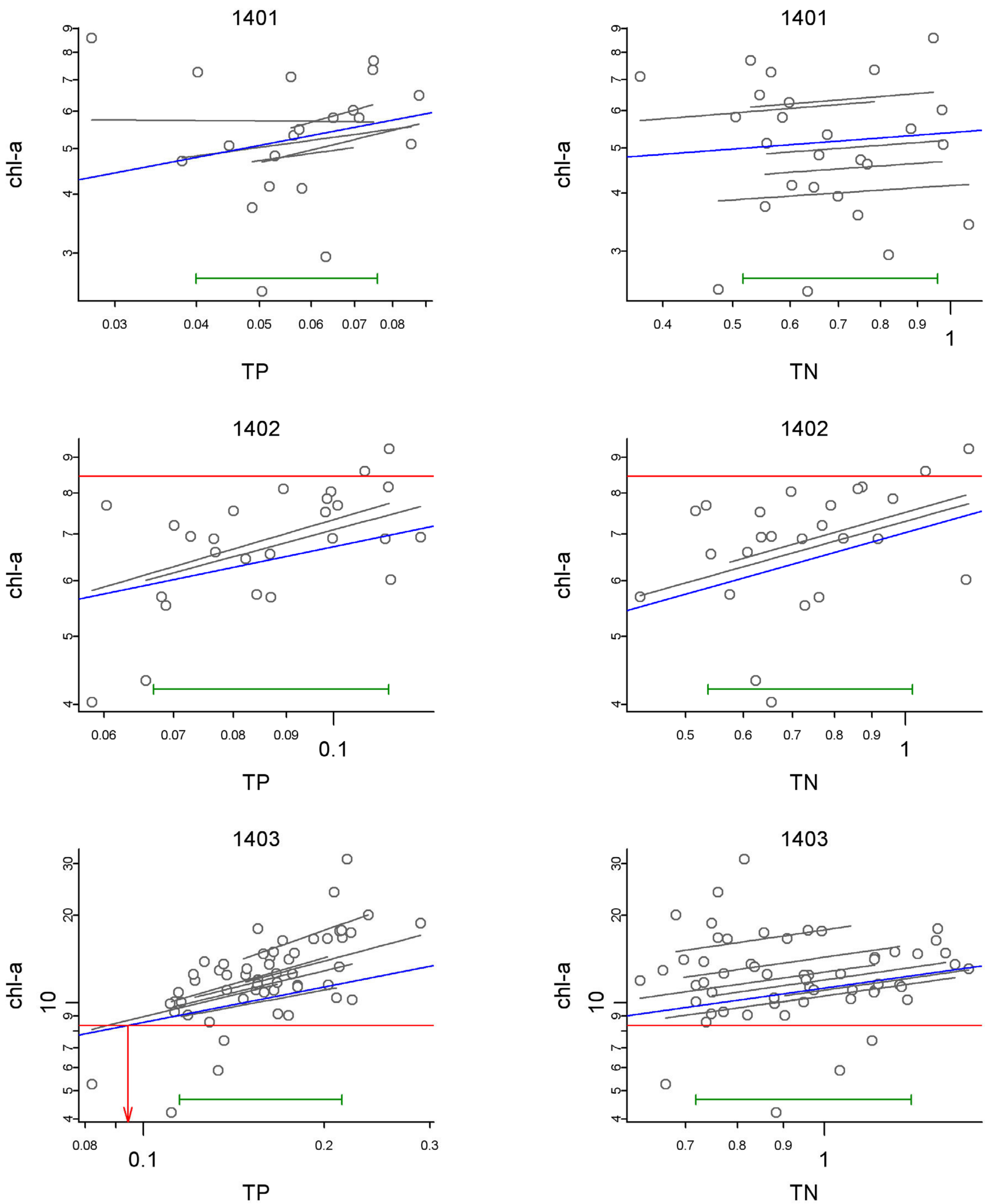

Figure 2-44. Relationships between TN, TP, and chl-a in St. Lucie Estuary. Open circles: observed annual average values of TN, TP, and chl-a, red horizontal line: chl-a criterion, red vertical arrow: TN and TP criteria associated with chl-a criterion, blue line: estimated segment-wide relationship between TN, TP, and chl-a, grey lines: estimated station-specific relationships between TN, TP, and chl-a, green line segment: 5th to 95th percentile range of observed annual geometric mean TN and TP concentrations.

Table 2-136. Summary of candidate TN and TP criteria for St. Lucie Estuary.
Asterisks indicate criteria based on the upper or lower bounds of available data.

Segment	TN (mg/L)	TP (mg/L)
1401	-	0.076*
1402	-	0.113*
1403	0.72*	0.115*

2.17.7. Application of Analyses for Proposed Numeric Nutrient Criteria

EPA evaluated various lines of evidence for deriving numeric nutrient criteria in the St. Lucie Estuary. Although data necessary to conduct statistical model analyses in St. Lucie River were available, EPA found that the relationship between nutrients and biological endpoints was not significant in all segments. EPA derived the proposed numeric nutrient criteria based on the mechanistic modeling results.

EPA evaluated three endpoints in the mechanistic modeling approach for St. Lucie River: (1) water clarity protective of historic depth of seagrasses, (2) chl-a concentrations associated with balanced phytoplankton biomass, and (3) DO concentrations sufficient to maintain aquatic life. EPA found that the 2002–2009 nutrient loads met all three endpoints. Therefore, the proposed criteria were derived to be protective of all three. The values under mechanistic modeling represent the 90th percentile annual geometric mean nutrient concentrations from the 2002–2009 modeled nutrient loads.

Analysis of available statistical data indicated that relationships estimated between chl-a concentrations and attenuation coefficient were not statistically significant, so no chl-a criteria were derived that were associated with the water clarity endpoint. In all segments, chl-a criteria were derived to meet the phytoplankton endpoint only. Relationships between TN, TP, and chl-a were used to calculate TN and TP concentrations associated with meeting chl-a criteria in each segment. As noted above, relationships estimated between TN and chl-a in segments 1401 and 1402 were not statistically significant, so no TN criteria were computed in those segments. In segments 1401 and 1402, TP criteria are based on the upper bound of observed data, while in segment 1403 these criteria are based on the lower bound of observed data.

EPA derived the proposed numeric nutrient criteria for St. Lucie in Table 2-137 based on the mechanistic modeling results. Criteria derived from statistical analysis corroborate the criteria derived from mechanistic model.

Table 2-137. Proposed and candidate numeric nutrient criteria for St. Lucie segments

		Proposed Criteria			Mechanistic Modeling			Statistical Modeling		
SEGMENT	SEGMENT ID	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
Lower St. Lucie	1401	0.58	0.045	5.3	0.58	0.045	5.3	-	0.076	9.4
Middle St. Lucie	1402	0.90	0.120	8.4	0.90	0.120	8.4	-	0.113	8.5
Upper St. Lucie	1403	1.22	0.197	8.9	1.22	0.197	8.9	0.72	0.115	8.4

2.17.8. Downstream Protective Values

In St. Lucie Estuary mechanistic models were applied to derive proposed DPVs for TN and TP shown in Table 2-138.

Table 2-138. Proposed DPVs for St. Lucie

Tributary[a]	LSPC Model Watershed ID	USGS Reach Code	Estuary Segment	TN (mg/L)	TP (mg/L)
Warner Creek	190051	03090206033664	1401	1.25	0.030
	190052	03090206018782	1402	1.37	0.030
Howard Creek	190055	03090206033620	1403	1.25	0.027
Winters Creek	190057	03090206033613	1403	1.19	0.027
	190059	03090206015188	1403	1.24	0.077
Frazier Creek	190066	03090206033669	1403	1.20	0.028
	190068	03090206018767	1403	1.15	0.040
Danworth Creek	190070	03090206033679	1403	0.81	0.054
St. Lucie River	190115	03090206033756	1401	1.22	0.026
	c23	03090206033652	1403	1.39	0.304
North Fork St. Lucie Canal (c24)	c24	03090206015188	1403	1.40	0.243
South Fork St. Lucie Canal (c44)	c44	03090206033715	1403	1.68	0.214

[a] Tributary names left blank are unnamed

2.17.9. References

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2.18. Indian River Lagoon

2.18.1. Proposed Numeric Nutrient Criteria

Proposed numeric nutrient criteria for TN, TP, and chl-a in Indian River Lagoon segments are summarized in Table 2-139.

Table 2-139. Proposed numeric nutrient criteria for Indian River Lagoon segments

Segment Name	Segment Number	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
Mosquito Lagoon	1501	1.18	0.078	7.5
Banana River	1502	1.17	0.036	5.7
Upper Indian River Lagoon	1503	1.63	0.074	9.2
Upper Central Indian River Lagoon	1504	1.33	0.076	9.2
Lower Central Indian River Lagoon	1505	1.12	0.117	8.7
Lower Indian River Lagoon	1506	0.49	0.037	4.0

2.18.2. General Characteristics

2.18.2.1. System Description

The Indian River Lagoon stretches approximately 155 mi (250 km) along Florida's eastern coast across six counties (FDEP 2008; SJRWMD 2007). For the purposes of describing this system, Indian River Lagoon will include Mosquito Lagoon, northern and central Indian River Lagoon, and the Banana River; the St. Lucie Estuary and southern Indian River Lagoon are described in a separate system description.[79] The Indian River Lagoon watershed drains approximately 2,284 mi^2 (5,915 km^2) (SJRWMD 2007). The northern and central sections of the Indian River Lagoon extend from southern Volusia and Brevard counties, through Indian River County, to the border of St. Lucie County in the south. From there, the Indian River Lagoon continues south along the coast toward the St. Lucie Estuary, which has three major inlets—Fort Pierce Inlet, St. Lucie Inlet, and Jupiter Inlet (FDEP 2008; SJRWMD 2007).

The annual average temperature measured from 1981 to 2010 at Melbourne in Indian River Lagoon is 72.4 °F (22.4 °C). Average maximum and minimum temperatures from the same time period are 81.9 °F (27.7 °C) and 62.8 °F (17.1 °C), respectively (Florida Climate Center 2010). Generally, the summer months are warm and humid in Indian River Lagoon. The winter months are mild, despite the occasional cold front passing through (Woodward-Clyde Consultants 1994a). In 2009, average annual rainfall measured at Kennedy Space Center and Cape Canaveral Air Force Station in the Indian River Lagoon watershed was 41.9 in (106.4 cm) (NASA Weather Office 2009). A pattern of wet and dry seasons occurs from May through October and November through April, respectively (FDEP 2008; Woodward-Clyde Consultants 1994a).

Barrier islands form the eastern border of the Indian River Lagoon. The lagoon morphology consists of a series of flat, low-gradient terraces separated by barrier beaches (FDEP 2008). The Indian River Lagoon surface sediment consists of sand and clay, dune sand, isolated peat deposits in lakes and marshes, and coquina shell debris close to the coast (FDEP 2008). The soil in the Indian River Lagoon is characterized by three main groups: (1) well-drained or excessively drained barrier island sands; (2) mainland knolls, flatwood soils, and coastal ridges (well-drained or excessively drained); and (3) swamp, marsh, slough, and hammock soils that are mostly poorly drained with high organic and clay content (Woodward-Clyde Consultants 1994a).

Land uses in the Indian River Lagoon are mainly wetlands, developed land, and agriculture (FDEP 2008). The population of Volusia, Brevard, and Indian River counties in the Indian River Lagoon watershed has grown 500 percent from fewer than 200,000 in the 1950s to more than one million people in 2008 (FDEP 2008). The population in these counties is projected to grow another 25 percent from 2010 to 2030 (FHDC 2011). Nearly 40 percent of the Mosquito Lagoon watershed is federally owned conservation land (Walters et al. 2001).

[79] The information presented in this system description was compiled to summarize local information pertaining to the proposed numeric nutrient criteria for this estuary. For more information on EPA's process of delineating, segmenting, and deriving numeric nutrient criteria for this estuary, please see Section 1.3.

The Indian River Lagoon is a series of interconnected lagoons whose circulation is influenced by their distance to an ocean inlet, winds, and amount of freshwater inflow; the lagoon has very limited exchange with ocean water, leading to higher salinities at some sites (FDEP 2009; SJRWMD 2007). The average tidal amplitude throughout both the Banana River and Indian River Lagoon is about 0.3 ft (0.1 m) (Steward et al. 2010a). Depth throughout the lagoon averages 3.9 ft (1.2 m) (SJRWMD 2007). Freshwater residence times vary greatly in relation to proximity to an ocean inlet, ranging from 18 days in central Indian River Lagoon to 226 days in Banana River. Residence times in Mosquito Lagoon range from 3.5 to 76 days, varying drastically with proximity to Ponce de Leon Inlet (Steward et al. 2010a). Three aquifers that underlie the landscape are the surficial aquifer, intermediate aquifer, and the Floridan aquifer (FDEP 2008). The Floridan aquifer in the Indian River Lagoon area has slightly elevated phosphorus concentrations (FDEP 2008).

2.18.2.2. Impaired Waters[80]

Twenty-three Class II and Class III marine WBIDs in the Indian River Lagoon area have been listed for a nutrient-related parameter on Florida's CWA section 303(d) list approved by EPA. Of the 23 WBIDs, seven are Class II WBIDs and 16 are Class III marine WBIDs. Of the seven Class II WBIDs, four are impaired for nutrients (WBIDs 2963A, 2963C, 5003B, and 5003D), two are impaired for DO (WBIDs 2924 and 2924B), and one is impaired for nutrients, chl-a, and DO (WBID 2963F). Of the 16 Class III marine WBIDs, three are impaired for DO (WBIDs 3085A, 3107, and 3129A), three are impaired for nutrients (WBIDs 2963E, 3044B, and 5003C), six are impaired for nutrients and DO (WBIDs 2963D, 3044A, 3057A, 3057B, 3057C, and 3135), and four are impaired for nutrients, chl-a, and DO (WBIDs 2963B, 3082, 3166, and 3190).[81]

Two nutrient-related TMDLs for Class II or Class III marine WBIDs exist in the Indian River Lagoon watershed. Those TMDLs are the final *Indian River Lagoon and Banana River Lagoon Nutrient and DO TMDL,* covering 13 Class II and III marine WBIDs, and the final *Northern and Central Indian River Lagoon and Banana River Lagoon Nutrients and DO TMDL,* covering 12 Class II and Class III marine WBIDs (2963A, 2963B, 2963C, 2963D, 3057A, 3057B, 3082, 3085A, 3129A, 3135, 5003C, 5003D). Note that eight of those WBIDs are covered by both TMDLs.[82]

[80] For more information about the data source, see Volume 1, Appendix A.
[81] The nutrient-related 303(d) list was a compilation of EPA-approved 303(d) listing information for nutrients, chl-a, and DO provided in three decision documents: September 2, 2009, Basin Groups 1, 2, and 5 EPA Decision Document (http://www.epa.gov/region4/water/tmdl/florida/documents/fl09303d_decisiondoc_090209.pdf);
May 13, 2010, Basin Group 3 EPA Decision Document (http://www.epa.gov/region4/water/tmdl/florida/documents/fl303d_%20partialapproval_decision_docs051410.pdf);
and the December 21, 2010, Basin Group 4 EPA Decision Document (http://www.epa.gov/region4/water/tmdl/florida/documents/group_4_final_dec_doc_and_partial_app_letter_12_21_10.pdf).
[82] TMDLs were identified in February 2011 by compiling nutrient-related draft/final TMDLs from the following three sources:
FDEP TMDL website (http://www.dep.state.fl.us/water/tmdl/index.htm)
EPA Region 4 website (http://www.epa.gov/region4/water/tmdl/florida/index.html)
EPA National WATERS expert query tool (http://www.epa.gov/waters/tmdl/expert_query.html).

2.18.2.3. Water Quality

The Indian River Lagoon has a long history of hydromodification from mosquito ditching, impounding marshes, and dredging activities (FDEP 2008; SJRWMD 2007). In addition, urban pressures from rapid population growth adjacent to the Indian River Lagoon have contributed substantial nutrient pollution loads and reduced clarity in the Indian River Lagoon (FDEP 2008; FHDC 2011).

NOAA's *Estuarine Eutrophication Survey* characterized the Indian River Lagoon as having elevated nutrient levels, especially during the summer. Chl-a concentrations over 20 µg/L and high turbidities (Secchi depth < 3.3 ft [1 m]) were reported, as were moderately high concentrations of nitrogen and phosphorus (TDN: 0.1–1.0 mg/L; TDP: 0.01–0.1 mg/L). Anoxia (DO = 0 mg/L) or hypoxia (DO ≤ 2.0 mg/L) during the spring and summer was estimated to occur over 10 to 25 percent of the lagoon, and nuisance and toxic algae during the summer were reported to occur periodically (NOAA 1996). In general, Mosquito Lagoon's nitrogen, chl-a, TSS, and turbidity double in concentration during the wet season. Those seasonal increases are said to occur naturally (Woodward-Clyde Consultants 1994b). Phosphorus concentrations reportedly declined in Mosquito Lagoon over the period of 1990–2004 (Winkler and Ceric 2006). Average TP values were 0.09 ± 0.05 mg/L (1991–1993 data) (Hall et al. 2001). Nutrient pollution is a concern in the southern area of the lagoon because of residence time (Steward et al. 2003). There are generally higher TN concentrations in the north, which decrease further south to the central Indian River Lagoon, with the exception of locations near the discharge of major drainage systems (SJRWMD 2007). Higher concentrations in the northern Indian River Lagoon and Banana River reflect a large standing pool of organic nitrogen (up to 95% of TN is organic). The TN-TP ratios in the Indian River Lagoon have generally been high (Phlips et al. 2002; SJRWMD 2007).

Throughout the Indian River Lagoon, DO concentrations tend to drop during the summer; however, grab sample data from 1988 to 1994 indicated that levels remained above 5 mg/L throughout the year. Between 1988 and 1994 there was a slight increasing trend in chl-a levels from north to south in the Indian River Lagoon, with the highest levels found in south-central Indian River Lagoon (Sigua et al. 2000). Spikes in chl-a have been associated with periodic algae blooms (SJRWMD 2007). Based on monitoring data between 1990 and 1999, chl-a levels in Mosquito Lagoon showed average annual concentrations of approximately 5–6 µg/L (Steward et al. 2003). Hall et al. (2001) noted a strong correlation with turbidity and TSS. Turbidity is generally lower in the Banana River and the north and central Indian River Lagoon (usually around 3 NTU) than in Mosquito Lagoon and the south-central Indian River Lagoon (SJRWMD 2007). Salinity throughout the Indian River Lagoon is variable and influenced heavily by both proximity to an ocean inlet and the amount of freshwater input in an area. In general, it has been above 20 PSU (Phlips et al. 2002; SJRWMD 2007). Color tends to follow spatial patterns of salinity, with increased color where increased freshwater inputs from relatively high color streams occur in southern Indian River Lagoon (SJRWMD 2007).

2.18.2.4. Biological Characteristics

More species are found in the Indian River Lagoon than in any other North American estuary. As of 2007, approximately 2,100 plant species and 2,200 animal species, including 685 fish species and 370 bird species are present in the Indian River Lagoon watershed (SJRWMD 2007).

The Indian River Lagoon is primarily characterized by biological communities composed of seagrass habitats, mangrove forests and salt marsh habitats, and an increasing amount of phytoplankton blooming in open water habitats. The Indian River Lagoon also contains about 27 percent of all Florida's east coast salt marshes. Primary species of mangroves in the Indian River Lagoon include the red mangrove, white mangrove, and black mangrove. Mangroves generally colonize the intertidal areas, whereas upland trees and shrubs colonize the interiors of the islands found in the lagoon (SJRWMD 2007).

Mosquito Lagoon is noted as having one of the greatest extents of salt marsh and mangrove forest in the Indian River Lagoon system (Woodward-Clyde Consultants 1994b). Approximately 1.94 mi^2 (5.02 km^2 [1,240 ac]) of tidal marsh and swamp are in the Mosquito Lagoon Aquatic Preserve system (FDEP 2009).

Seven species of seagrasses grow in the Indian River Lagoon; the most common are turtle grass (*Thalassia testudinum*), manatee grass (*Syringodium filiforme*), and shoal grass (*Halodule wrightii*) (Steward and Green 2007). Seagrasses in Indian River Lagoon covered an estimated total of 112 mi^2 (290 km^2 [71,646 ac]) in 2007 (FFWCC 2011). Between 1996 and 2007 overall seagrass acreage in the Indian River Lagoon increased; for example there has been an increase of 9 percent between 2005 and 2007 (FFWCC 2011; SJRWMD 2007). However, some other areas have experienced tremendous seagrass acreage loss (Steward and Green 2007).

Oyster reefs and hard clams are present but not widespread in the Indian River Lagoon system (SJRWMD 2007; Steward et al. 2003; Woodward-Clyde Consultants 1994b). Areas of limited tidal flushing, such as the Titusville, Melbourne, and Cocoa Beach areas, have the highest mean phytoplankton standing crops (Phlips et al. 2002). Blooms of several species of dinoflagellates, diatoms, and cyanobacteria are common and relatively frequent throughout the Indian River Lagoon (Badylak and Phlips 2004). From 2006 to 2009, 24 HAB species were observed in the Indian River Lagoon, including 8 species at bloom levels (Phlips et al. 2011). The toxic form of the dinoflagellate *Pyrodinium* var. *bahamense* has been widespread throughout the Indian River Lagoon (Phlips et al. 2006; Phlips et al. 2011). Throughout the Indian River Lagoon, unattached benthic macroalgae appears in aggregations of variable size; the dominant species is *Gracilaria* spp., which composes over 90 percent of the macroalgal biomass in Indian River Lagoon (Virnstein and Carbonara 1985).

Common commercial and recreational invertebrate species in the Indian River Lagoon include hard clams (*Mercenaria mercenaria*), rock shrimp (*Sicyonia brevirostris*), white shrimp (*Penaeus setiferus*), oysters (*Crassostrea virginica*), and blue crabs (*Callinectes sapidus*) (FDEP 2008). Landings of shellfish have generally declined since the 1980s. Approximately 700 benthic macroinvertebrate species have been recorded in the Indian River Lagoon, predominantly amphipods, crustaceans, gastropods, and polychaete worms (Woodward-Clyde Consultants 1994b).

Commercial fisheries in the Indian River Lagoon have been decreasing; annual commercial landings in 2005 were nearly half those from 1959 to 1962 (IRLNEP 1996). Catch data from the past 30 years show a severe decline in catches for species such as snook and spotted seatrout (SJRWMD 2007). The Florida manatee and bottlenose dolphin are among the 30 mammal species living in the Indian River Lagoon (Woodward-Clyde Consultants 1994b).

For a more detailed summary of this water body, see Volume 1, Appendix A.

2.18.3. Data Used

Several data sources specific to Indian River Lagoon were used in addition to those sources described in Section 1.4.3, as summarized in Table 2-140.

Table 2-140. Data sources specific to Indian River Lagoon models

Data	Source	Location Used
FDEP Level III Florida Land Use	South Florida Water Management District (SFWMD 2011b)	Indian River watershed model
Flow gaging stations	South Florida Water Management District (SFWMD 2011a)	Indian River watershed model
Arc Hydro Enhanced Database watershed boundaries	South Florida Water Management District (SFWMD 2011a)	Indian River watershed model
Hydrologic group soils data	St. Johns River Water Management District (SJRWMD 2011)	Indian River watershed model

2.18.4. Segmentation

The GIS isohaline analysis and geomorphological structure of the Indian River Lagoon yielded six segments with salinity decreasing from the upper to the lower portion of the lagoon. Figure 2-45 shows the segments for Indian River Lagoon.

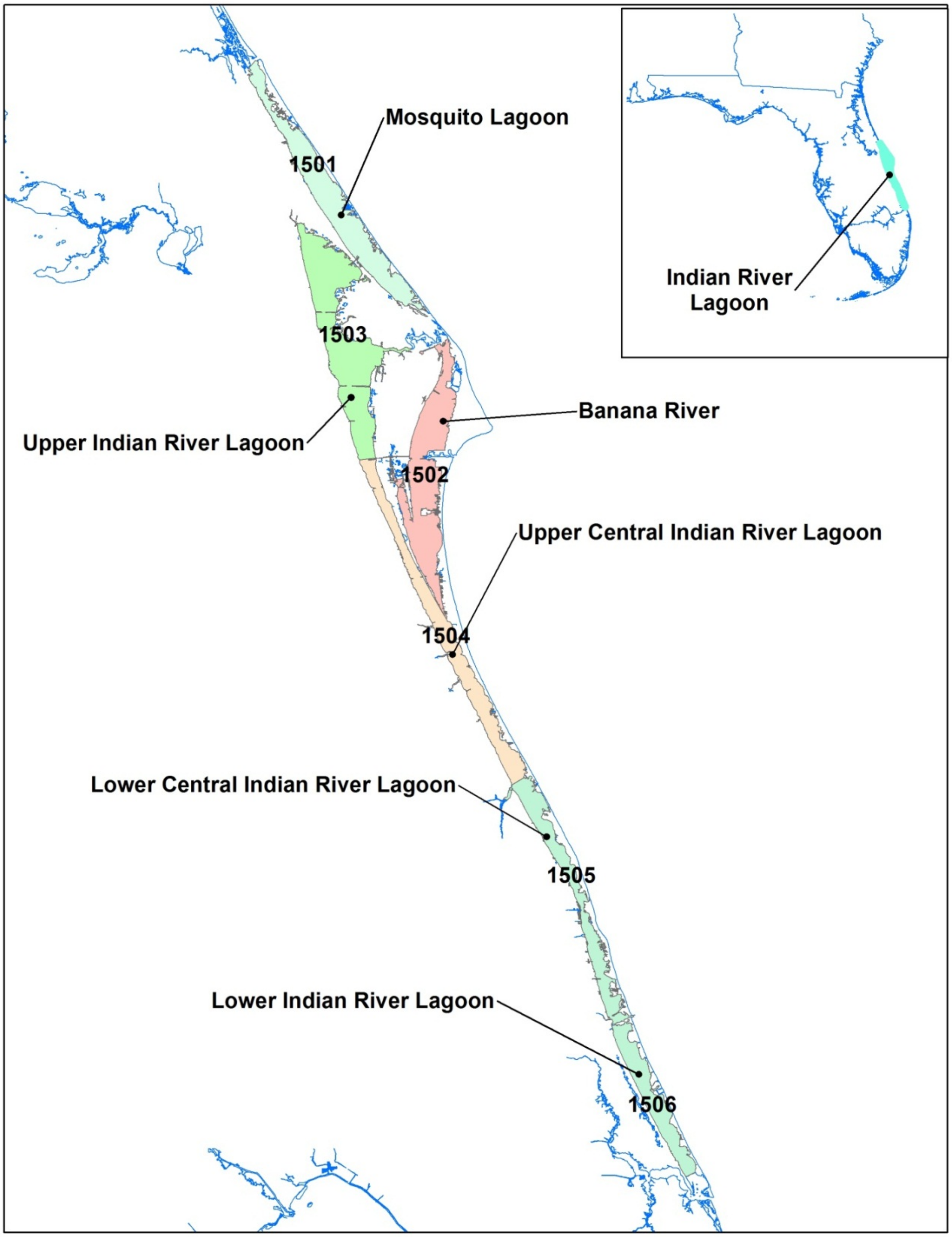

Figure 2-45. Results of Indian River Lagoon segmentation

2.18.5. Water Quality Targets

2.18.5.1. Seagrass Depth and Water Clarity Targets

Seagrass depth of colonization (measured as Z_c) and water clarity (measured as K_d) targets were established for six estuary segments in Indian River Lagoon (Table 2-141) based on seagrass coverage maps for 1943 and 2009 (SJRWMD 2012). For segment 1506, seagrass data were available for 2009 but not 1943. Average CDOM in Indian River Lagoon did not reduce photosynthetically active radiation at the depth of colonization target to less than 20 percent of surface irradiance in any of the segments. Depth of colonization targets were set to the maximum value, which occurred in 2009 for 1502, 1503, and 1505 and in 1943 for segment 1504. Since the maximum value for many segments occurred in 2009, the 2009 depth of colonization was used as the target for segment 1506.

Table 2-141. Indian River Lagoon seagrass depth and water clarity targets by segment

Estuary Segment	Depth of Colonization (Z_c) Target (m)	Light Attenuation Coefficient (K_d) Target (1/m)
1501	1.1	1.46
1502	1.3	1.24
1503	1.1	1.46
1504	1.3	1.24
1505	1.1	1.46
1506	1.7	0.95

2.18.5.2. Chlorophyll a Target

To prevent nuisance algal blooms and protect the estuary's designated uses, chl-a levels must not exceed 20 µg/L more than 10 percent of the time. The rationale for that target is provided in Section 1.2.2.

2.18.5.3. Dissolved Oxygen Targets

On the basis of the rationale that sufficient DO is necessary to protect aquatic life, as described in Section 1.2.3, the following DO targets were established:

- Minimum allowable DO of 4.0 mg/L as a water column average for each estuary segment 90 percent of the time over the simulation's time span
- Daily average DO of 5.0 mg/L as a water column average for each estuary segment 90 percent of the time over the simulation's time span
- Minimum 3-hour average DO of 1.5 mg/L in the bottom two layers for each estuary segment over the simulation's time span

2.18.6. Results of Analyses

2.18.6.1. Mechanistic Model Analysis

Indian River Lagoon was not evaluated using mechanistic model analysis.

2.18.6.2. Statistical Model Analysis

Analysis of data using statistical models provided estimates of relationships between chl-a and light attenuation coefficient in Indian River Lagoon (Figure 2-46). When candidate criteria values for chl-a based on stressor-response were less than the lower bound of the range of data (segments 1502 and 1506), the candidate criteria were based on the lower bound of the data. Similarly, when candidate criteria values for chl-a based on stressor-response were greater than the upper bound (segment 1501), the candidate criteria were based on the upper bound of the data. The estimated relationships between light attenuation coefficient and chl-a in segment 1503 was not statistically significant, so no chl-a criteria associated with water clarity were derived for this segment.

EPA also estimated relationships between bloom frequency and TN, TP, and chl-a, and derived candidate criteria values based on these relationships (Figure 2-47). In all segments, proposed candidate criteria are protective of both the water clarity and the bloom frequency endpoints (Table 2-142).

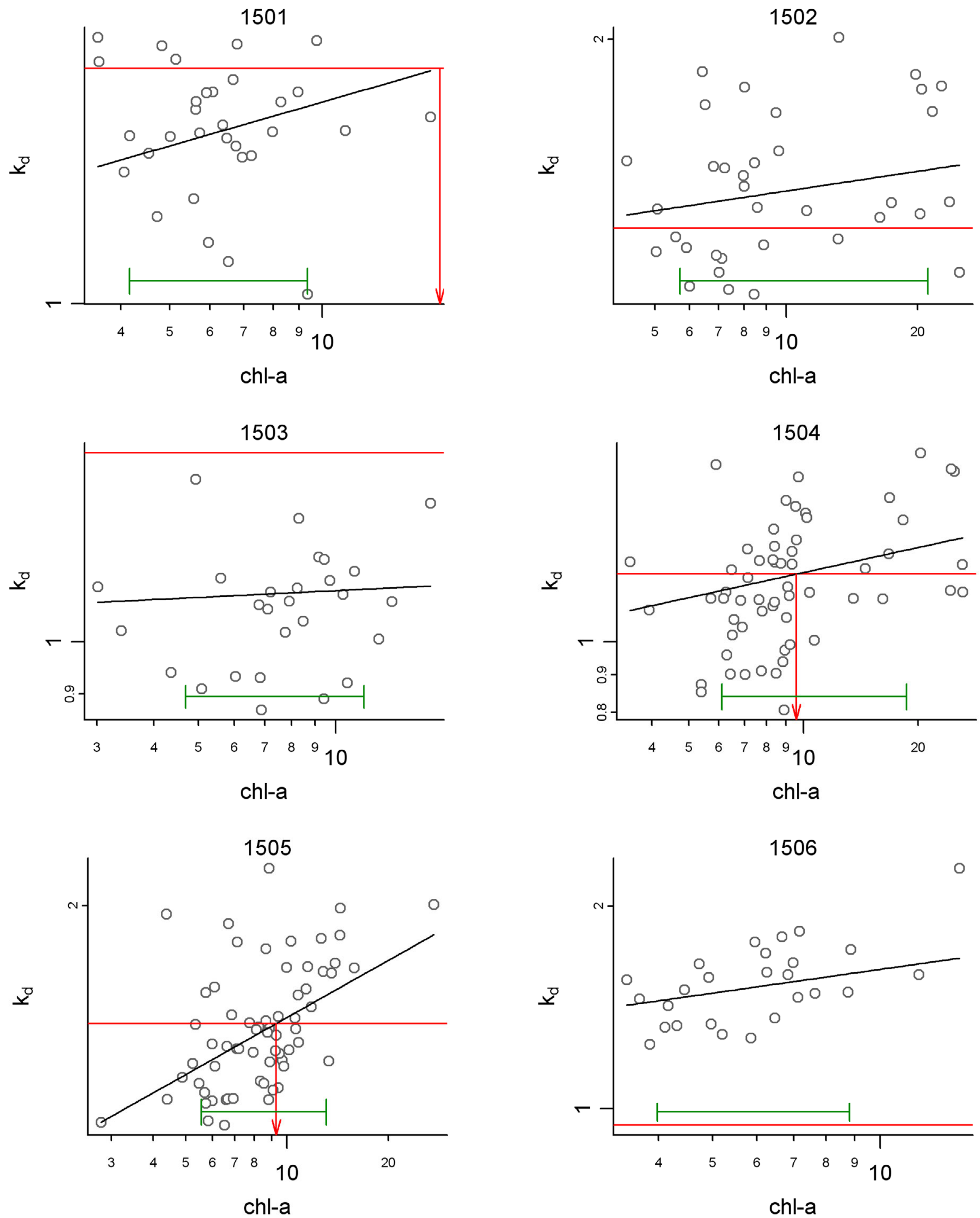

Figure 2-46. Relationships between chl-a and corrected K_d in Indian River Lagoon. Red horizontal line shows K_d corresponding with water clarity target. Red vertical arrow show annual geometric mean chl-a concentration predicted to be associated with K_d target. Green line segment shows the 5th to 95th percentile range of observed geometric mean values for chl-a. Open circles: observed values of K_d corrected for the effects of turbidity and color.

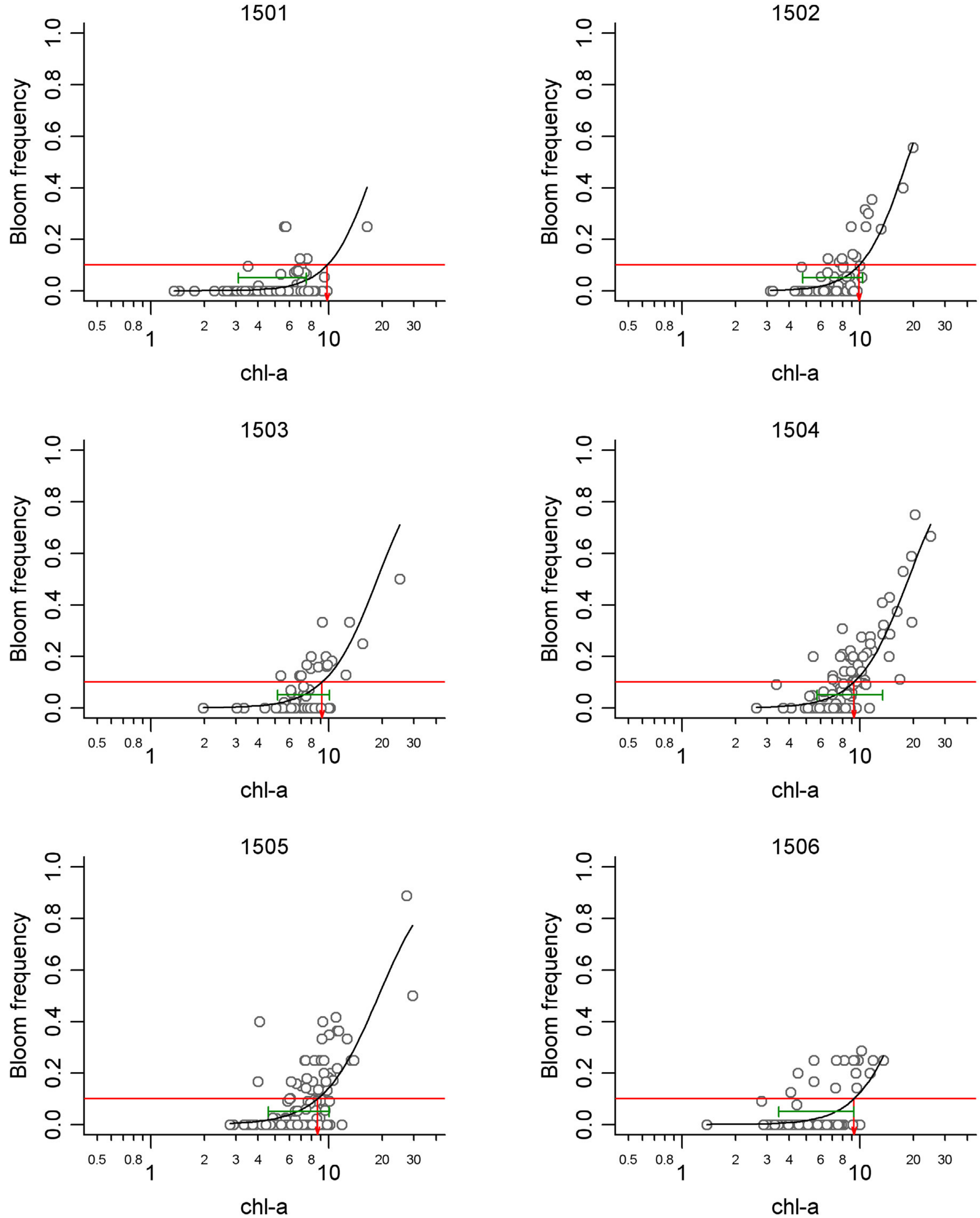

Figure 2-47. Estimates of annual geometric chl-a concentrations associated with bloom frequency in Indian River Lagoon. Red horizontal line: bloom frequency of 0.1, red vertical arrow: annual geometric mean chl-a concentration associated with 0.1 bloom frequency, green line segment: 5th to 95th percentile range of observed data.

Table 2-142. Summary of candidate chl-a criteria in Indian River Lagoon. Values with asterisks indicate that the predicted candidate criterion was greater than the upper bound of chl-a values, or less than the lower bound of chl-a values used in estimating the empirical relationship, so listed criterion is based on the upper or lower bound of the data.

Segment	Chl-a (clarity)	Chl-a (bloom)	Chl-a (final)
1501	9.4*	7.5*	7.5
1502	5.7*	9.9	5.7
1503	-	9.2	9.2
1504	9.6	9.2	9.2
1505	9.3	8.7	8.7
1506	4.0*	9.2*	4.0

Statistical relationships between TN and chl-a, and TP and chl-a were used to derived TN and TP criteria that are consistent with the chl-a criteria (Figure 2-48 and Figure 2-49). Here, as with the light attenuation coefficient—chl-a relationships, when candidate criteria derived from the stressor-response relationships are outside the range of observed data, they are based on the upper or lower bound of the observed data (Table 2-143).

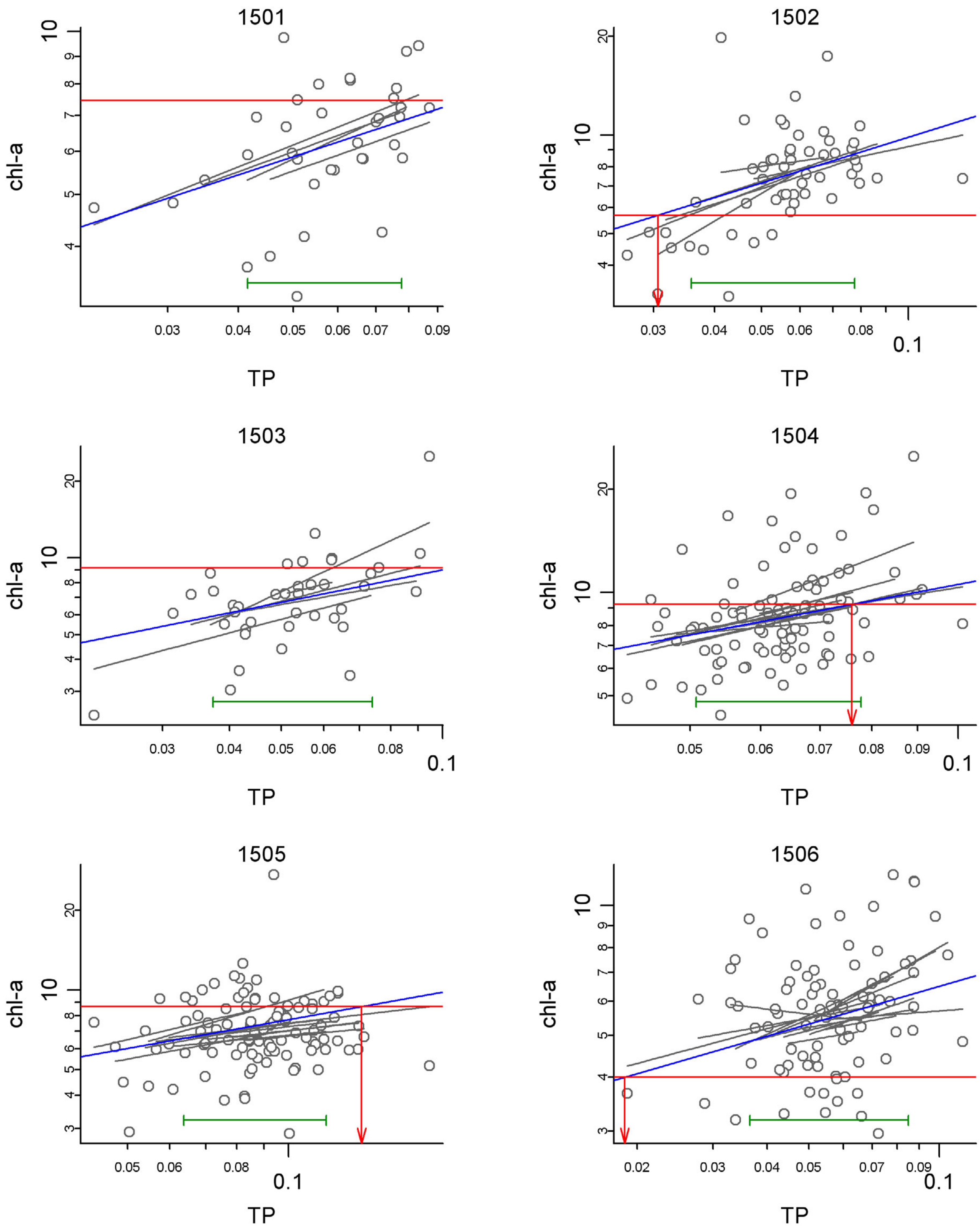

Figure 2-48. Relationships between TP and chl-a in Indian River Lagoon. Open circles: observed annual average values of TP and chl-a, red horizontal line: chl-a criterion, red vertical arrow: TP criterion associated with chl-a criterion, blue line: estimated segment-wide relationship between TP and chl-a, grey lines: estimated station-specific relationships between TP and chl-a, green line segment: 5th to 95th percentile range of observed annual geometric mean TP concentrations.

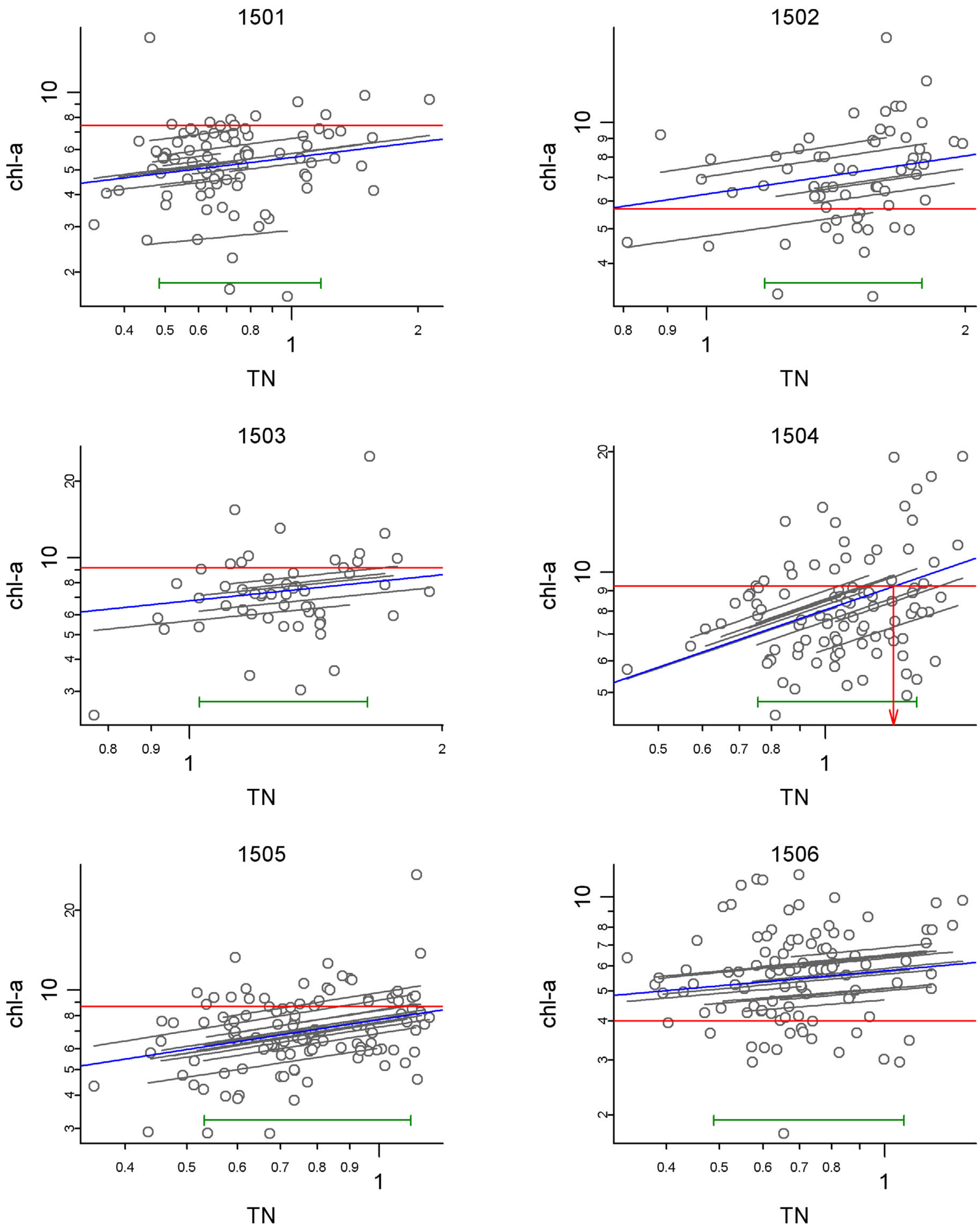

Figure 2-49. Relationships between TN and chl-a in Indian River Lagoon. Open circles: observed annual average values of TN and chl-a, red horizontal line: chl-a criterion, red vertical arrow: TN criterion associated with chl-a criterion, blue line: estimated segment-wide relationship between TN and chl-a, grey lines: estimated station-specific relationships between TN and chl-a, green line segment: 5th to 95th percentile range of observed annual geometric mean TN concentrations.

Table 2-143. Summary of candidate TN and TP criteria for Indian River Lagoon. Values with asterisks indicate that the predicted candidate criterion was greater than the upper bound of observed values, or less than the lower bound of observed values used in estimating the empirical relationship, so listed criterion is based on the upper or lower bound of the data.

Segment	TN (mg/L)	TP (mg/L)
1501	1.18*	0.078*
1502	1.17*	0.036*
1503	1.63*	0.074*
1504	1.33	0.076
1505	1.12*	0.117*
1506	0.49*	0.037*

2.18.7. Application of Analyses for Proposed Numeric Nutrient Criteria

In Indian River Lagoon there was sufficient data to conduct a statistical model for every segment. EPA evaluated two endpoints in the statistical modeling approach for Indian River Lagoon: (1) water clarity protective of historic depth of seagrasses, and (2) chl-a concentrations associated with balanced phytoplankton biomass. Through evaluating the observed data EPA found that, in some segments, the TN, TP, and chl-a concentrations associated with achieving the water quality targets for the biological endpoints were greater than the range of TN, TP, or chl-a concentrations observed in the available data for Indian River Lagoon. For these segments, EPA is proposing to set numeric criteria derived from statistically modeled relationships at the upper bound of the distribution of available data instead of deriving criteria outside the range of data observations (see Appendix B). Similarly, for concentrations less than the lower bound of the range of observed data EPA is proposing to set numeric criteria at the lower bound of the observed data. This approach defines criteria values that maintain balanced natural populations of aquatic flora and fauna within the limits of available data.

Proposed numeric nutrient criteria for TN, TP, and chl-a in Indian River Lagoon segments are summarized in Table 2-144.

Table 2-144. Proposed and candidate numeric nutrient criteria for Indian River Lagoon segments

		Proposed Criteria			Statistical Modeling		
SEGMENT	SEGMENT ID	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
Mosquito Lagoon	1501	1.18	0.078	7.5	1.18	0.078	7.5
Banana River	1502	1.17	0.036	5.7	1.17	0.036	5.7
Upper Indian River Lagoon	1503	1.63	0.074	9.2	1.63	0.074	9.2
Upper Central Indian River Lagoon	1504	1.33	0.076	9.2	1.33	0.076	9.2
Lower Central Indian River Lagoon	1505	1.12	0.117	8.7	1.12	0.117	8.7
Lower Indian River Lagoon	1506	0.49	0.037	4.0	0.49	0.037	4.0

2.18.8. Downstream Protective Values

In Indian River Lagoon mechanistic models were not available to derive DPVs for TN and TP. In lieu of the preferred approach a mixing/dilution model was applied to calculate the allowable freshwater TN and TP load. This mixing/dilution model assumes that TN and TP loads in freshwater mix conservatively with saline seawater. Using this assumption, the model predicts freshwater concentrations necessary to achieve proposed nutrient criteria in each segment (Figure 2-50 and Figure 2-51). Proposed DPVs are shown in Table 2-145.

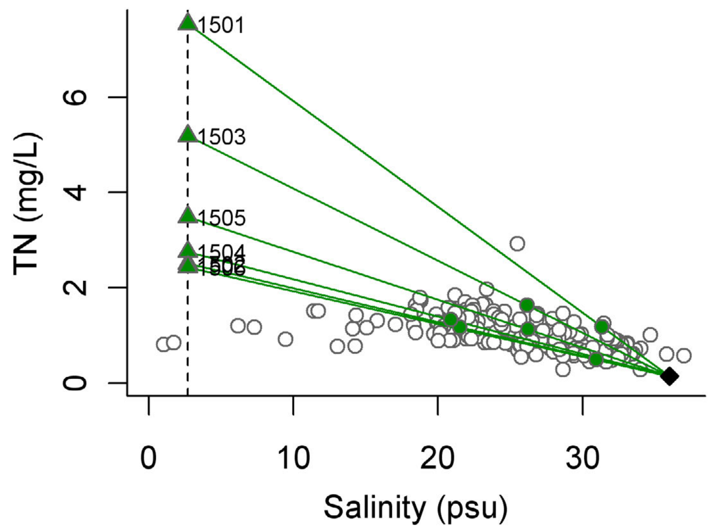

Figure 2-50. Calculation of TN DPVs for Indian River Lagoon. Black diamond shows seawater conditions, filled green circles show proposed TN criteria values for each segment, and green triangles show calculated DPVs. Open circles show observed long-term station average TN concentrations and salinities.

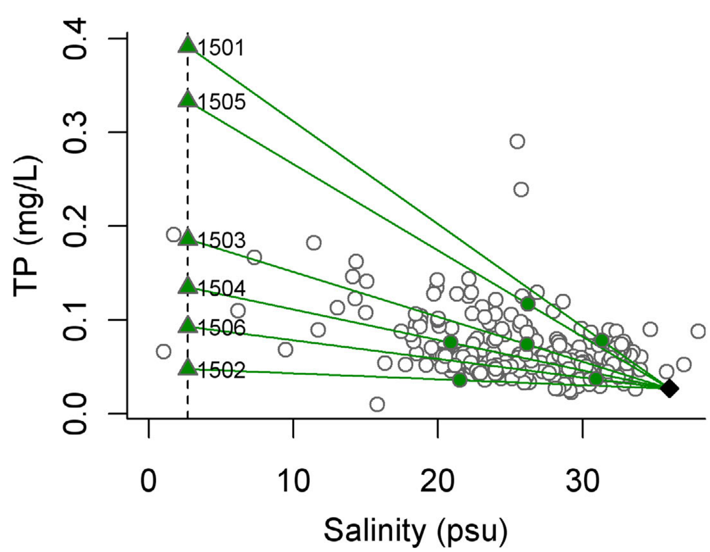

Figure 2-51. Calculation of TP DPVs for Indian River Lagoon. Black diamond shows seawater conditions, filled green circles show proposed TP criteria values for each segment, and green triangles show calculated DPVs. Open circles show observed long-term station average TP concentrations and salinities.

Table 2-145. Proposed DPVs for Indian River Lagoon

Segment	DPV (TP) (mg/L)	DPV (TN) (mg/L)
1501	0.391	7.53
1502	0.048	2.51
1503	0.186	5.19
1504	0.135	2.75
1505	0.333	3.48
1506	0.093	2.43

2.18.9. Alternate Analyses

2.18.9.1. Indian River and Banana River

The St. Johns River Water Management District (SJRWMD) took a multi-method approach to derive both loading- and concentration-based numeric nutrient criteria for segments in the Indian River and Banana River—the two seagrass dominated systems in SJRWMD's jurisdiction.[83] Analyses were conducted to address nutrient levels and their relationship to the following ecological endpoints: (1) seagrass depth limit, light requirements, chl-a, macroalgae, and epiphytes; and (2) general trophic state and primary production. The analyses and lines of evidence include seagrass depth-limit regression models, a reference segment-year (RSY) method, sub-lagoon seagrass light attenuation models (optical models), two general nutrient models that pertain to estuaries (described in more detail below). An analysis specific to HABs was also conducted, primarily on *Pyrodinium bahamenese* var. *bahamense*, a resident species of the Indian River and Banana River system that has increased in bloom frequency and poses serious health threats to humans. The HAB analysis was performed to better understand the relationship between bloom intensity and nutrient loads, as well as to potentially establish thresholds that limit bloom harm. Segmentation was based on the spatial variation in water quality attributes such as depth limit targets per segment and watershed loadings (TN, TP, TSS) per segment, resulting in 15 final segments. Analyses were performed using field data from the Indian River and Banana River water quality and seagrass monitoring programs run by SJRWMD (period of record from 1989 to 2007) and data from the University of Florida for the HAB analyses (period of record from 1997 to 2009).

The final SJRWMD-proposed TN and TP criteria concentrations were calculated on the basis of the RSY method. Two criteria magnitudes were proposed, one as an annual median and the other as a wet season (June–October) monthly maximum. Alternate criteria were also proposed. These were derived using a convergence of the concentrations calculated by the RSY method and general models. Targets for chl-a were presented as a range of values established by using the optical model approach and the reference segment approach (Table 2-146).

[83] See Volume 1, Appendix G

Table 2-146. Annual mean turbidity (NTU), chl-a concentration (µg/L), and color (PCU) from both the optical model and RSY method. Adapted from Steward et al. (2010a)

	Turbidity (NTU)		Chl-a (µg/L)		Color (PCU)		TN (mg/L)		TP (mg/L)	
Method	**Optical**	**RSY**	**Optical**	**RSY**	**Optical**	**RSY**	**Median**	**Mean**	**Median**	**Mean**
Banana River	4.1	2.9	4.7	2.7	19.6	15	1.32	1.34	0.029	0.032
North Indian River Lagoon	3.3	3.5	2.5	4.6	14.5	20	1.33	1.3	0.045	0.051
North Central Indian River Lagoon	N/A	2.5	N/A	3	N/A	15	0.82	0.84	0.041	0.049
Sebastian Inlet	3.4	2.8	3.6	2.2	24.9	15	0.6	0.63	0.047	0.052
South Central Indian River Lagoon	2.9	3.6	1.7	2.7	22.3	20	0.68	0.72	0.075	0.085

Seagrass depth-limit regression models. As a first line of evidence, numeric criteria were derived using loading limits from a 2009 TMDL (Section 62-304.520, F.A.C.). To derive TN and TP criteria, a regression model initially formulated for the TMDL served as a foundation to create TN/TP loading–seagrass-depth-limit regression models for each of the major sub-lagoons. This parent model regression, was used to back-predict areal TN and TP loading limits relative to a 10 percent departure from full restoration seagrass depth-limit targets (consistent with Florida's transparency standard that states the depth of compensation point for photosynthetic activity under background conditions shall not be decreased by more than 10 percent). Available mapping years for seagrass GIS coverage were 1943, 1986, 1989, 1992, 1994, 1996, 1999. For Indian River and Banana River, the reference period was defined as the aggregate of segment years that achieved the TMDL seagrass targets. Depth points at dredged areas were excluded from the analysis, and all remaining year GIS coverages were layered. The depth points closest to the union boundary for all years were used to calculate the depth limit. The median depth limits per segment year (1943, 1996, 1999, 2001) were used as the percent departure from the depth-limit target, which were later regressed on the log-transformed TP or TN loading rate. The median depth limit targets for each segment ranged from 1.18 to 1.81 m. The total sub-lagoon nutrient loading limits were then calculated by multiplying model-predicted areal loading limits for each sub-lagoon by the area of that sub-lagoon.

RSY method. The second method used data from reference segments that met the desired depth of colonization targets established in the TMDL analyses. Seagrass mapping for segments that met the state-established depth threshold were identified and then data for chl-a, color, TN, and TP were aggregated over 6, 12, and 18 months for each segment to determine a month-based median for each parameter. According to Steward et al. (2010a), those monthly time periods were selected because they all precede or overlap the seagrass growing season during the mapping year. It was assumed that the results of the reference segment year method would indicate the water quality conditions needed to meet the state-designated depth limit. The annual medians for the 12-month period and the 90th percentile from the reference segment year method were listed in the final report.

Seagrass light attenuation or optical models. The third approach relied on several multivariate geometric mean function regression (GMFR) optical models that were built for each of the sub-lagoons using data from 1996 to 2007. The sub-lagoons in this analysis are the Banana River

Lagoon, North Indian River Lagoon, and Central Indian River Lagoon (divided into Sebastian and South Central reaches). An optical model is still being developed for the North Central reach. This type of regression was chosen to both minimize measurement error among the variables and incorporate more than one explanatory variable (i.e., color, turbidity, and chl-a) into the analysis. However, the GMFR model (Draper and Yang 1997) is unable to provide calculations on confidence intervals; therefore, results must be compared to other lines of evidence. Data sets were checked to ensure assumptions of parametric models were met. Outliers were identified and removed, constituting less than 5 percent of the data points for each sub-lagoon. To predict the median chl-a, the RSY median values (50^{th} percentile) for color and turbidity and the light attenuation coefficient target were entered into the regression equation and solved for chl-a. To arrive at the maximum monthly chl-a value, the 10^{th} percentile of turbidity and color were used in the regression equation.

General nutrient models. The fourth approach applied the two general empirical models of Steward and Lowe (2010) and Dettmann (2001) to data specific to the Indian River Lagoon and Banana River Lagoon. Both models were used to predict nutrient loading and concentration limits, as well as trophic state. The models were created using data from Florida systems that had both established TMDLs and where upper mesotrophy was a water management goal. The Steward and Lowe (2010) model, also called the Florida TN and TP model, predicts a nutrient load limit as a function of water residence time or hydraulic loading for mesotrophic systems. Using the Florida TN model, the sub-lagoons were plotted to determine their status along a trophic continuum. The Florida TP model was mathematically analogous to the Vollenweider (1976) TP critical loading model and, thus, was used to predict TP. However, the Florida TN loading model was unable to be adapted to the nitrogen analog of the Vollenweider (1976) model to predict TN concentrations. Because only the Dettmann (2001) model accounted for denitrification, it was used to predict the concentration of TN on the basis of TN loading limits; however, the model could not be applied to land-bound segments where a strong longitudinal gradient existed or TN concentrations were unavailable because of a lack of data. Where the Dettmann (2001) model could not be used to predict TN concentrations, a TN–TP ratio for the given sub-lagoon was applied to the TP limit to acquire TN limits. That was the case for both Banana River Lagoon and North Indian River Lagoon.

Loading data (lb/ac/yr) per sub-lagoon were determined from calibrated watershed models, including the Pollutant Load Screening Model and HSPF. Loading data were also concurrent with seagrass mapping years (1943, 1996, 1999, 2001). Total external TN and TP loading limits for the sub-lagoons were normalized to estuarine surface water area, and then paired with their respective hydraulic loadings. To determine the degree of mesotrophy, TN and TP areal loading limits per segment were log transformed and plotted against hydraulic loading established by Steward and Lowe (2010).

Pyrodinum bahamense analysis. The fifth approach relied on the relationship between HAB occurrence and TP concentrations. Targets for chl-a were presented as a range of values established using both the optical model approach and the reference segment approach. Proposed TN and TP loading criteria were based on the loading limits determined in the TMDL analyses. Primary proposed TN and TP criteria concentrations were calculated using the reference segment method. Alternate criteria were derived using a convergence of the concentrations calculated by the reference segment method and general models. Two criteria magnitudes were proposed, one for an annual median and the other for a wet season (June–October) monthly maximum. The regression analysis indicated that TP concentrations ranging from 0.020 to 0.025 mg/L have the potential to support *P. bahamense* blooms of at least 100 cells/mL. However, a threat level for saxitoxin has yet to be identified, and TP concentrations based on the RSY method might not be low enough to prevent *P. bahamense* blooms.

Documentation of SJRWMD-proposed numeric nutrient criteria and methods for their derivation can be found in Appendix G.

The different lines of evidence and results from the various analyses were contrasted to determine the degree of agreement found between the predicted nutrient concentration limits. Both empirical models were found to support the reference period results. A strong correlation was also found between reference period and model predicted concentrations of TN and TP, although TP lacked the one-to-one correspondence seen for TN. Moreover, it was determined that the results from both the RSY method and general models were in fair to good agreement (~10 to 24 percent absolute difference based on the RSY medians; 0 to 24 percent difference based on the RSY means) and both could be used collectively to determine protective nutrient criteria for seagrasses. For instance, both methods indicated that chl-a should be maintained below 5 µg/L in Banana River Lagoon and North Indian River Lagoon and below 4 µg/L in central Indian River Lagoon to attain depth of colonization target limits. It was concluded that the external total nutrient loading estimates can serve as reasonably accurate loading limits to protect the subestuaries in the Indian River and Banana River.

2.18.9.2. Mosquito Lagoon

SJRWMD submitted a document to EPA detailing proposed methods to derive numeric nutrient criteria for Mosquito Lagoon including the resulting criteria (Table 2-147). SJRWMD divided Mosquito Lagoon into three segments based on the distinct surrounding land use, hydrodynamics, habitat, and water quality of each segment. The northern segment extends from Ponce Inlet to Edgewater, the central segment from Edgewater to Oak Hill, and the southern segment from Oak Hill to Merritt Island. Documentation of SJRWMD-proposed numeric nutrient criteria and methods for their derivation can be found in Volume 1, Appendix G.

Table 2-147. Mosquito Lagoon annual median chl-a targets and model-predicted concentration limits for TN and TP (Steward et al. 2010b)

Estuary Segment	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)
ML1	2.9	0.48	0.065
ML2	2.3	0.52	0.035
ML3-4	2.2	0.8	0.021

For Mosquito Lagoon, SJRWMD proposed a suite of five approaches to develop a weight-of-evidence by which numeric criteria can be developed. The approaches were based on one of three relationships: (1) the link between nutrients, phytoplankton growth (as shown by chl-a), and the trophic state of a system; (2) the link between nutrients, phytoplankton growth (as shown by chl-a), the effects of phytoplankton on light attenuation in the water column, and the light requirements of seagrasses; or (3) the connection between TP and HAB occurrence.

The first and primary approach uses a reference period from 2004 to 2008 to calculate annual median and maximum wet-season medians of TN, TP, and chl-a. The reference period was selected because of the low TN, TP, and chl-a observed during that period; the rainfall amounts during that period were representative of typical rainfall over time; and the Trophic State Index (TSI) value for that period was less than 50, which was deemed to be desirable (mesotrophy to oligo-mesotrophy). The second approach draws on an optical model linking chl-a to previously established water clarity targets as a way to predict annual median chl-a in southern Mosquito Lagoon that would be protective of seagrasses and serve as a basis for criteria derivation. The third approach derived a TP level that corresponds to minimum bloom levels of the dinoflagellate *Pyrodinium bahamense*, the common HAB species seen primarily in the southern Lagoon. A fourth line of evidence applied to the Mosquito Lagoon uses multivariate GMFR models to relate TN and TP to chl-a on an annual basis and during the wet season. Targets for chl-a were set on the basis of the reference period mentioned above for the north and central segments and the optical model for the southern segments. The final method was based on two general nutrient models of Steward and Lowe (2010) and Dettmann (2001). The reference method was used to derive the TN, TP, and chl-a criteria for Mosquito Lagoon, with the other four methods providing supporting evidence. Two criteria magnitudes for TN, TP, and chl-a were presented, one an annual median value and the other a wet-season (July to September) median value.

2.18.10. References

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2.19. Halifax River

2.19.1. Proposed Numeric Nutrient Criteria

Proposed numeric nutrient criteria for TN, TP, and chl-a in Halifax River segments are summarized in Table 2-148.

Table 2-148. Proposed numeric nutrient criteria for Halifax River segments

Segment Name	Segment Number	TN (mg/L)	TP (mg/L)	Chl-a (μg/L)
Upper Halifax River	1601	0.75	0.243	9.4
Lower Halifax River	1602	0.63	0.167	9.6

2.19.2. General Characteristics

2.19.2.1. System Description

The Halifax River Estuary is a long, narrow, shallow, marshy tidal lagoon on Florida's northeast coast in Volusia and Flagler counties (Haydt and Frazel 2003). The river basin drains approximately 340 mi^2 (880 km^2), with a length of about 23 mi (37 km), an average width of 0.6 mi (0.9 km), and mean depths ranging between 4.6 ft (1.4 m) and 5.6 ft (1.7 m) (Steward et al. 2010).

Major freshwater inputs into the estuary come from Rose Bay and three tributaries (Bulow Creek, the Tomoka River, and Spruce Creek) (Haydt and Frazel 2003). The Halifax River Estuary has two tidal nodes: one is between South Daytona and Daytona Beach, the other is just south of High Bridge, east of the confluence with Bulow Creek. The Halifax River Estuary is commonly divided into North Halifax River Estuary and South Halifax River Estuary on the basis of hydrodynamic characteristics, habitat distinctions, catchment land use differences, and spatial water quality data differences (Steward et al. 2010).[84] The Halifax River Estuary is connected to the Guana, Tolomato, Matanzas, and Pellicer rivers to the north through the Intracoastal Waterway and the Indian River Lagoon to the south through Mosquito Lagoon (FDEP 2005, 2008b). Ponce de Leon Inlet is the dominant source of tidal exchange and flushing in the Halifax River Estuary, while the Matanzas Inlet makes a relatively small contribution that is confined to the far northern portion of the Halifax River Estuary. A majority of the Halifax River Estuary Basin has been altered by mosquito impoundments, residential development, and silviculture (FDEP 2008b; Haydt and Frazel 2003; Militello and Zarillo 2000).

The Halifax River Estuary is between the barrier beaches to the east and the drainage divide formed by the Talbot Terrace to the west (FDEP 2008b). The Halifax River Estuary's basin morphology consists of a sequence of flat, low-gradient terraces, divided by barrier beaches (Scholl et al. 1980). The basin is covered by undifferentiated sediments consisting of sand and clay, dune sand, isolated peat deposits, and coquina shell debris, with over 80 percent of the total area made up of poorly drained wetland soils (FDEP 2008b; SJRWMD 2011). The elevation change over the Halifax River Estuary Basin is very slight (the average elevation is 23 ft [7 m]; the range is 5–66 ft [1.5–20 m]). The flow in the basin is primarily directed by tides and wind (Haydt and Frazel 2003; SJRWMD 2011).

The Halifax River Estuary Basin is characterized as subtropical with an average high temperature of 81 ˚F (27 ˚C) in the summer and 61.5 ˚F (16.4 ˚C) in the winter (USGS 1985; Volusia County 2011). Average annual rainfall in Daytona Beach is 48.4 in (123 cm) (1951–1980 data). Nearly half the annual rainfall totals occur during the wet season, from June to September (USGS 1985).

The Halifax River Estuary Basin is projected to experience intense population growth. Population in Flagler and Volusia counties is expected to increase by 89 and 21 percent, respectively, between 2010 and 2030 (FHDC 2011). Urbanized areas surrounding the Halifax River Estuary include Daytona Beach (2010 population: 61,005), Port Orange (56,048), Ormond Beach (38,137), and New Smyrna Beach (22,644) (US Census Bureau 2012).

Wetlands and open water are the dominant land use in both North Halifax River Estuary (37%) and South Halifax River Estuary (35%). Range and forest have similar land coverage, composing 36 and 33 percent of North and South Halifax River Estuary, respectively. Urban and residential compose approximately 22 and 24 percent of North and South Halifax River Estuary, respectively. Land use proportions in both basins are nearly identical; however, the North Halifax River Estuary is settled more densely than the South Halifax River Estuary (Steward et al. 2010).

[84] The information presented in this system description was compiled to summarize local information pertaining to the proposed numeric nutrient criteria for this estuary. For more information on EPA's process of delineating, segmenting, and deriving numeric nutrient criteria for this estuary, please see Section 1.3.

2.19.2.2. Impaired Waters[85]

Seven Class III marine WBIDs in the Halifax River Estuary have been listed for a nutrient-related parameter on Florida's CWA section 303(d) list approved by EPA. Of the seven Class III marine WBIDs, two are impaired for DO (WBIDs 2620 and 2634A), one is impaired for nutrients (WBID 2363A), one is impaired for nutrients and chl-a (WBID 2670), one is impaired for nutrients and DO (WBID 2672), and two are impaired for nutrients, chl-a, and DO (WBIDs 2674A and 2678). No Class II WBIDs with nutrient-related impairments are documented for this area.[86] Two final nutrient-related TMDLs for Class II or Class III marine WBIDs exist in the Halifax River Estuary watershed. The TMDLs are the *Halifax River Nutrients and DO TMDL,* covering Class III marine WBIDs 2963A and 2963B (listed as water body IDs FL-0063 and FL-0064 in documentation, which was extrapolated out to current WBIDs) and the *Spruce Creek Dissolved Oxygen and Nutrient TMDL,* covering Class III marine WBID 2674A.[87]

2.19.2.3. Water Quality

A water quality survey conducted in the Halifax River Estuary in 1976 showed an average DO concentration of 5.3 mg/L (range 0–8.7 mg/L) (Scholl et al. 1980). Data collected close to the Beach Memorial Bridge from 1996 to 2011 (n = 122) showed a mean DO concentration of 6.7 mg/L (range 3.9–11.1 mg/L) (SJRWMD 2011). DO levels in Spruce Creek failed to meet the Florida DO water quality standard[88] 23.4 percent of the time between 1999 and 2006 (FDEP 2008a). A pronounced seasonality is apparent in chl-a concentrations, with peak concentrations observed during summer (wet season) and coincident with periods of elevated TN, TP, and temperature. Steward et al. (2010) asserted that temperature is the major factor regulating chl-a concentration variations between wet and dry seasons (Steward et al. 2010).

Turbidity varies geographically, with higher turbidities in the North Halifax River Estuary (Steward et al. 2010). Turbidity in the Halifax River Estuary is considered higher than typically found in other estuaries, such as Indian River Lagoon to the south; a median turbidity of 8.8 NTU (range 1.8–46.0 NTU) was measured at a sampling location 100 ft (30 m) north of the Beach Memorial Bridge between 1990 and 2008 (SJRWMD 2011).

[85] For more information about the data source, see Volume 1, Appendix A.
[86] The nutrient-related 303(d) list was a compilation of EPA-approved 303(d) listing information for nutrients, chl-a, and DO provided in three decision documents: September 2, 2009, Basin Groups 1, 2, and 5 EPA Decision Document (http://www.epa.gov/region4/water/tmdl/florida/documents/fl09303d_decisiondoc_090209.pdf);
May 13, 2010, Basin Group 3 EPA Decision Document
(http://www.epa.gov/region4/water/tmdl/florida/documents/fl303d_%20partialapproval_decision_docs051410.pdf);
and the December 21, 2010, Basin Group 4 EPA Decision Document
(http://www.epa.gov/region4/water/tmdl/florida/documents/group_4_final_dec_doc_and_partial_app_letter_12_21_10.pdf).
[87] TMDLs were identified in February 2011 by compiling nutrient-related draft/final TMDLs from the following three sources:
FDEP TMDL website (http://www.dep.state.fl.us/water/tmdl/index.htm)
EPA Region 4 website (http://www.epa.gov/region4/water/tmdl/florida/index.html)
EPA National WATERS expert query tool (http://www.epa.gov/waters/tmdl/expert_query.html).
[88] The Florida DO standard states that DO "shall not average less than 5.0 (mg/L) in a 24-hour period and shall never be less than 4.0 (mg/L). Normal daily and seasonal fluctuations above these levels shall be maintained" (Section 62-302.530, F.A.C.).

Long-term average TN data from 1992 to 2008 in the Halifax River Estuary indicate a decrease in TN concentrations in the early 1990s, followed by a relatively stable period from 1996 to 2008. Overall, the long-term annual median from 1992 to 2008 was 0.58 mg/L in North Halifax River Estuary and 0.30 mg/L in South Halifax River Estuary. During the stable period from 2000 to 2008, annual medians of 0.61 mg/L TN in the North Halifax River Estuary and 0.39 mg/L in the South Halifax River Estuary were found (Steward et al. 2010).

No long-term trend in TP concentrations was seen in the Halifax River Estuary from 1992 to 2008. Annual average TP values over the period ranged 0.14–0.17 mg/L (Steward et al. 2010).

The shallower North Halifax River Estuary had a lower and more uniform salinity (mean salinity around 20 PSU) than South Halifax River Estuary (mean salinity ranging from 20 to 30 PSU from north to south) between 1992 and 2008 (Steward et al. 2010). According to a 2004 SJRWMD study, from 1991 to 1999 salinity throughout the Halifax River averaged 23.46 PSU and average salinity in Rose Bay was 19.40 PSU. In the tributaries to the Halifax River, average salinities were lower: 11.99 PSU in Spruce Creek, 11.23 PSU in Bulow Creek, and 5.17 PSU in the Tomoka River (Miller 2004).

TSI values, another indicator of the biological health of a water body, in the Halifax River Estuary have decreased from the 1990s to the 2000s (FSU 1993; Steward et al. 2010).

2.19.2.4. Biological Characteristics

The Halifax River Estuary Basin is primarily characterized by biological communities composed of tidal marshes and oyster reefs (Paperno et al. 2001; Steward et al. 2010). The Tomoka Marsh Aquatic Preserve in North Halifax River Estuary is an important nursery for fish, shrimp, and crabs, as well as commercial and recreational fishing activities. More than 50 species of fish have been found in the preserve, as well as manatees, marine turtles, and bottlenose dolphins (FDEP 2011). Dominant species in salt marshes of the Halifax River Estuary include cordgrasses (*Spartina* spp.), rushes (*Juncus* spp.), glassworts (*Salicornia* spp.), and saltwort (*Batis maritima*) (Paperno et al. 2001). Spoil islands and wetland communities are also present as a result of mosquito ditching (FFWCC 1999; Paperno et al. 2001). Seagrass is largely absent throughout the Halifax River Estuary because of a combination of high turbidity and dredge and fill activities (Paperno et al. 2001; Steward et al. 2010).

The presence of oysters in the Halifax River Estuary is primarily affected by dredging (Steward et al. 2010).

Fish species and abundance are distributed throughout the basin depending on salinity gradients. In the northern Tomoka area, the fish community is characterized by freshwater fish species and seasonal recruitment of juvenile commercial shrimp (*Penaeidae* spp.) and drum (*Sciaenidae* spp.). The southern section near the Ponce de Leon Inlet area is characterized by species associated with marine waters such as Atlantic thread herring (*Opisthonema oglinum*) and scaled sardine (*Harengula jaguana*) (FFWCC 1999; Paperno et al. 2001; USGS 2010). A 2002–2004 USGS survey of the greater Florida Northern Coastal Basin found 157 fish species, which include bay anchovy (*Anchoa mitchilli*), spot (*Leiostomus xanthurus*), Atlantic croaker

(*Micropogonias undulaues*), mojarras (*Eucinostomus* spp.), and striped anchovy (*Anchoa hepsetus*) (USGS 2010).

Pyrodinium bahamense var. *bahamense* was observed in one (in August 2001) out of 48 samples collected over 2 years of observation (Phlips et al. 2006). Isolated algal blooms have been observed in at least one of the major freshwater streams feeding the Halifax River Estuary (Bulow Creek) (Bacchus and Barile 2005). Forty-one accounts of fish kills resulting from algae or red tide were reported from 2000 to 2010 in Flagler and Volusia counties. Overall, 78 percent of those reports were the result of a red tide occurrence in 2007 (FFWCC 2011; Frazel 2009).

Limited information is available on the health of macroinvertebrates in the Halifax River Estuary. In a 2000 study, three of the four sampling sites in the Halifax River Estuary region exhibited moderate species diversity scores (Shannon-Weiner Species Diversity Index)[89] and moderate tolerant taxa dominance. The sampling site at Tomoko River at Old Dixie Highway had very high invertebrate density, with small, shrimp-like crustaceans called tanaids (*Halmyrapseudes bahamensis*) composing nearly 89 percent of the total individuals over multiple collections (SJRWMD 2000).

For a more detailed summary of this water body, see Volume 1, Appendix A.

2.19.3. Data Used

No additional data sources specific to the Halifax River estuary were available; see general data sources described in Section 1.4.3.

2.19.4. Segmentation

The GIS isohaline yielded two segments for Halifax River: Upper Halifax River and Lower Halifax River. Figure 2-52 shows the two Halifax River segments.

[89] Shannon-Weiner Species Diversity Index is "a calculated index value expressing the degree of species diversity in a given sample or group of samples. The calculation is influenced by both the number of species present as well as the evenness of abundance among the species. Values generally range from 0 to 5, with values at the high end of the range indicating high species diversity" (SJRWMD 2000).

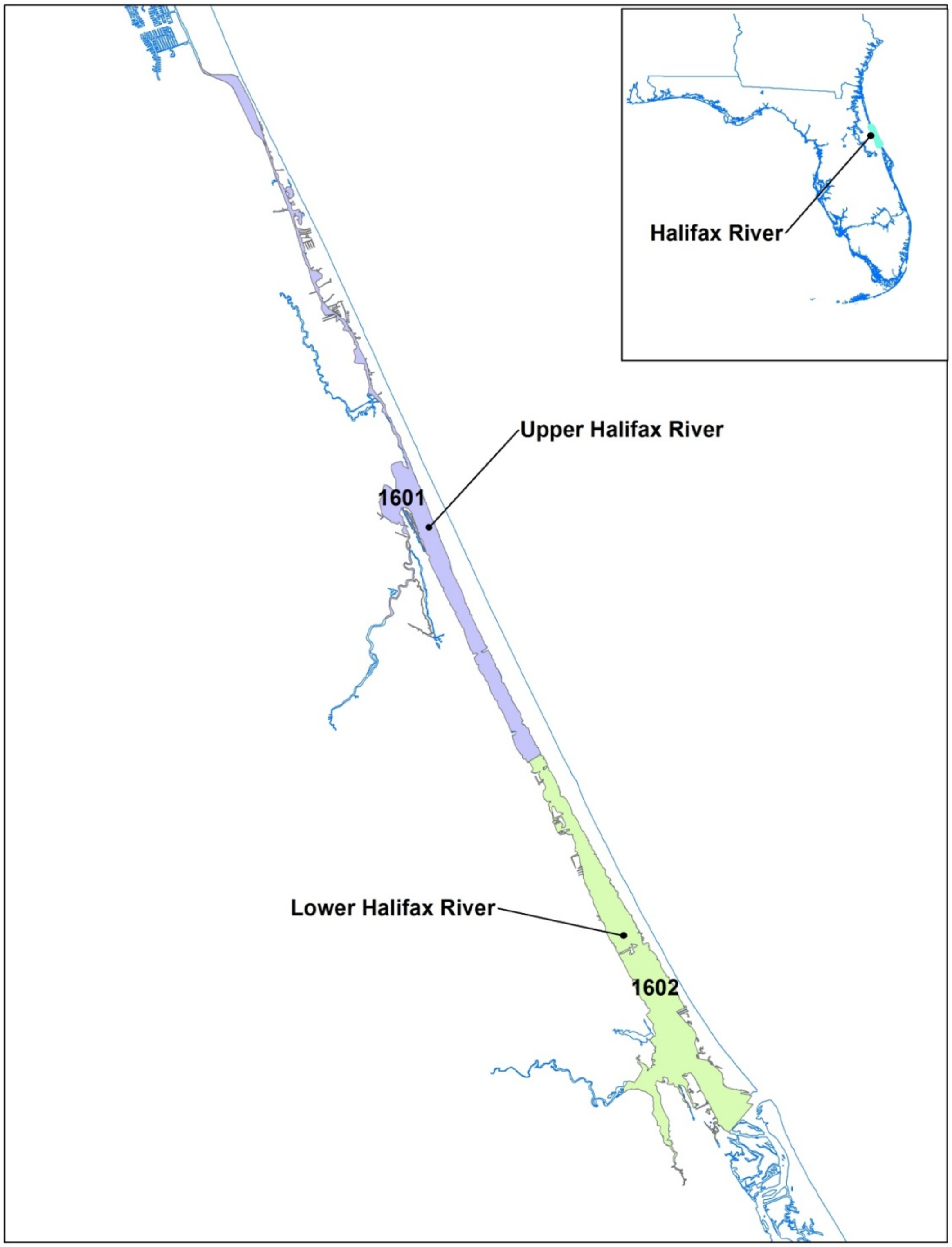

Figure 2-52. Results of Halifax River segmentation

2.19.5. Water Quality Targets

2.19.5.1. Seagrass Depth and Water Clarity Targets

Seagrass is not historically present in the Halifax River Estuary.

2.19.5.2. Chlorophyll a Target

To prevent nuisance algal blooms and protect the estuary's designated uses, chl-a levels must not exceed 20 µg/L more than 10 percent of the time. The rationale for that target is provided in Section 1.2.2.

2.19.5.3. Dissolved Oxygen Targets

On the basis of the rationale that sufficient DO is necessary to protect aquatic life, as described in Section 1.2.3, the following DO targets were established:

- Minimum allowable DO of 4.0 mg/L as a water column average for each estuary segment 90 percent of the time over the simulation's time span
- Daily average DO of 5.0 mg/L as a water column average for each estuary segment 90 percent of the time over the simulation's time span
- Minimum 3-hour average DO of 1.5 mg/L in the bottom two layers for each estuary segment over the simulation's time span

2.19.6. Results of Analyses

2.19.6.1. Mechanistic Model Analysis

Halifax River was not evaluated using mechanistic model analysis.

2.19.6.2. Statistical Model Analysis

Analysis of available empirical data using statistical models provided relationships between bloom frequency chl-a for Halifax River segments. No endpoints for water clarity were available. Candidate chl-a criteria were derived (9.4 µg/L and 9.6 µg/L, for segments 1601 and 1602 respectively), such that the predicted bloom frequency was 10 percent (Figure 2-53).

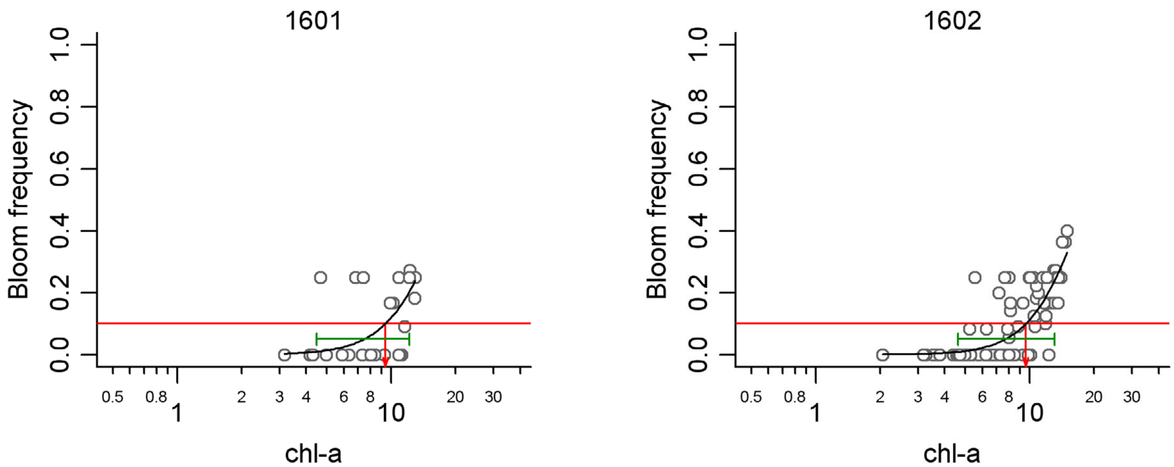

Figure 2-53. Modeled relationship between bloom frequency and chl-a in the Halifax River. Solid black line: modeled mean relationship. Open circles: observed annual geometric means. Red horizontal line: 10 percent bloom frequency endpoint. Green line segment: 5th to 95th percentile range of observed data.

Increasing concentrations of annual geometric mean TN and TP were associated with increased concentrations of chl-a, and segment-wide estimates of these relationships were used to derive TN and TP criteria values (Figure 2-54). In segment 1601, TN and TP concentrations associated with the chl-a criterion were both greater than the upper bound of observed values, so candidate criteria are based on the upper bound of observed annual geometric means (0.75 mg/L for TN, and 0.243 mg/L for TP). In segment 1602, proposed criteria are based on concentrations that are associated with the chl-a criterion (0.63 mg/L for TN and 0.167 mg/L for TP).

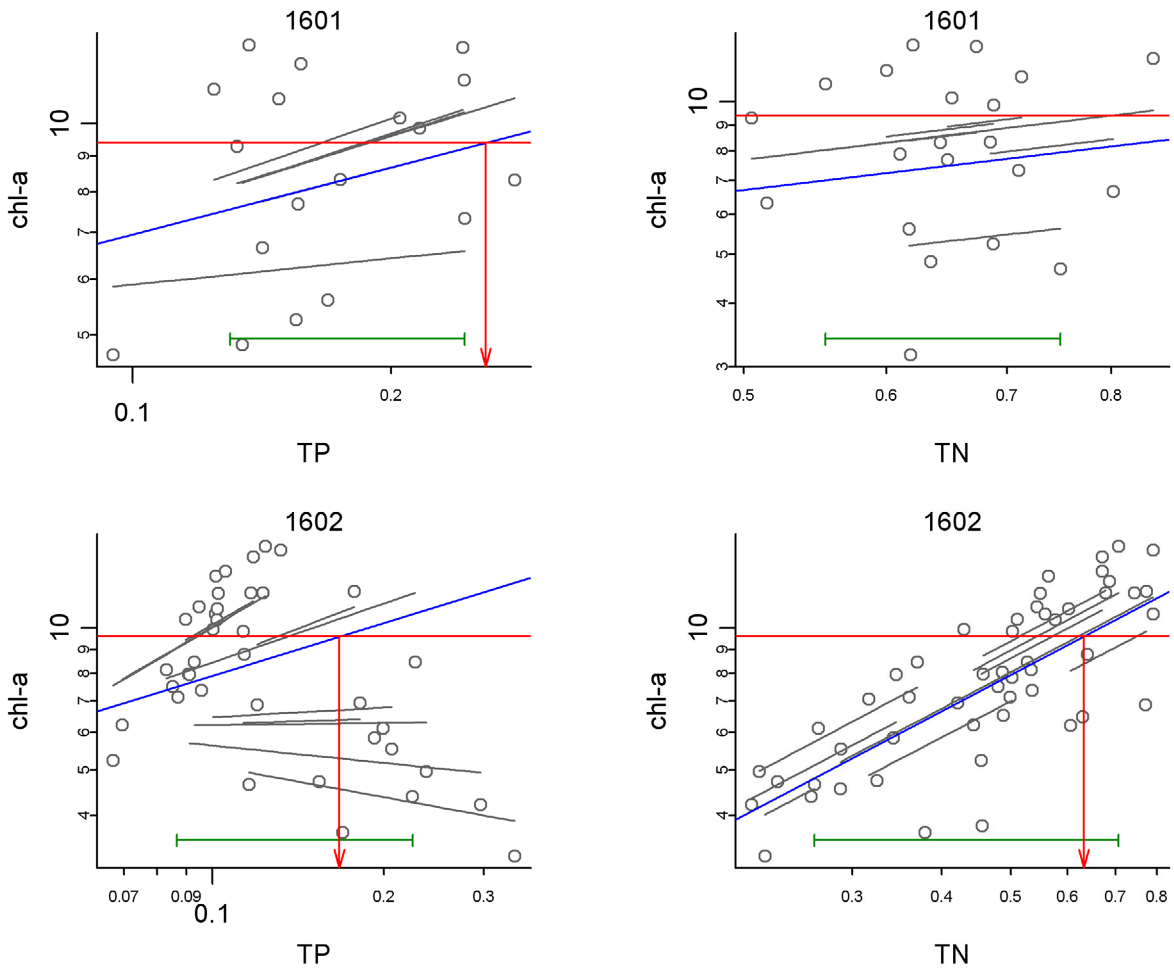

Figure 2-54. Relationships between TN, TP, and chl-a in the Halifax River. Open circles: observed annual average values of TN, TP, and chl-a, red horizontal line: chl-a criterion, red vertical arrow: TN and TP criteria associated with chl-a criterion, blue line: estimated segment-wide relationship between TN, TP, and chl-a, grey lines: estimated station-specific relationships between TN, TP, and chl-a, green line segment: 5th to 95th percentile range of observed annual geometric mean TN and TP concentrations.

2.19.7. Application of Analyses for Proposed Numeric Nutrient Criteria

In Halifax River Estuary data were sufficient to use statistical modeling analysis analyses as the primary line of evidence when deriving criteria. Seagrass has not been historically present in Halifax River, so EPA evaluated the following endpoint in the statistical modeling approach: (1) chl-a concentrations associated with balanced phytoplankton biomass. Through evaluating the observed data, EPA found that, in some segments, the TN or TP concentrations associated with achieving the chl-a target was greater than the range of TN or TP concentrations observed in the available data for Halifax River Estuary. For these segments, EPA is proposing to set numeric nutrient criteria derived from statistically modeled relationships at the upper bound of the distribution of available data instead of deriving criteria outside the range of data observations (see Appendix B). This approach defines criteria values that maintain balanced natural populations of aquatic flora and fauna within the limits of available data.

Proposed numeric nutrient criteria for TN, TP, and chl-a in Halifax River segments are summarized in Table 2-149.

Table 2-149. Proposed and candidate numeric nutrient criteria for Halifax River segments

		Proposed Criteria			Statistical Modeling		
SEGMENT	SEGMENT ID	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
Upper Halifax River	1601	0.75	0.243	9.4	0.75	0.243	9.4
Lower Halifax River	1602	0.63	0.167	9.6	0.63	0.167	9.6

2.19.8. Downstream Protective Values

In the Halifax River Estuary mechanistic models were not available to derive DPVs for TN and TP. In lieu of the preferred approach, a dilution/mixing model was applied to calculate the allowable freshwater TN and TP load. This mixing/dilution model assumes that TN and TP loads in freshwater mix conservatively with saline seawater. Using this assumption, the model predicts freshwater concentrations necessary to achieve proposed nutrient criteria in each segment (Figure 2-55 and Figure 2-56). DPVs are tabulated in Table 2-150.

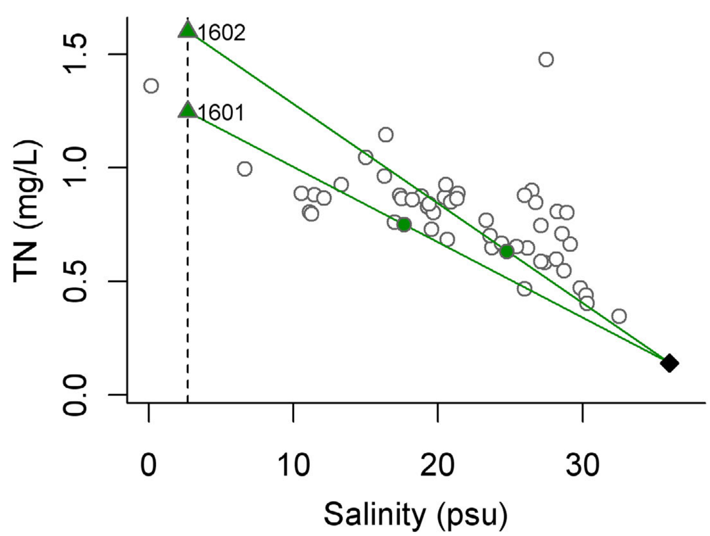

Figure 2-55. Calculation of TN DPVs for the Halifax River. Black diamond shows seawater conditions, filled green circles show proposed TN criteria values for each segment, and green triangles show calculated DPVs. Open circles show observed long-term station average TN concentrations and salinities.

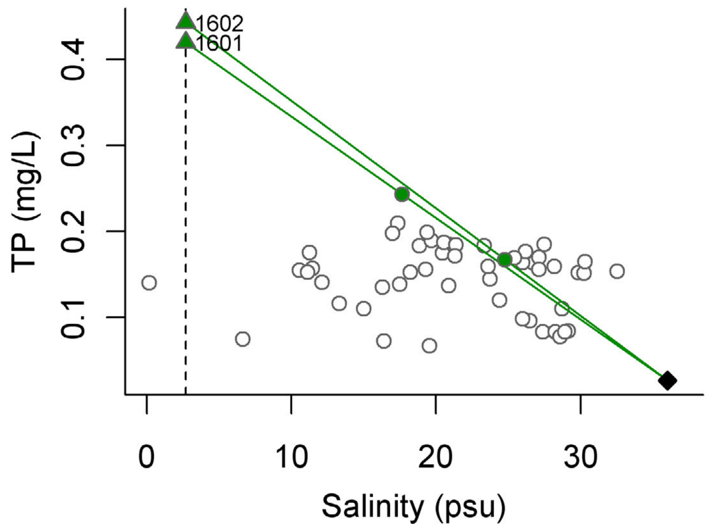

Figure 2-56. Calculation of TP DPVs for the Halifax River. Black diamond shows seawater conditions, filled green circles show proposed TP criteria values for each segment, and green triangles show calculated DPVs. Open circles show observed long-term station average TP concentrations and salinities.

Table 2-150. DPVs for TN and TP for the Halifax River

Segment	DPV (TP) (mg/L)	DPV (TN) (mg/L)
1601	0.419	1.25
1602	0.442	1.60

2.19.9. Alternate Analysis

The SJRWMD submitted documents to EPA suggesting approaches to derive numeric criteria for the Halifax River Estuary (aside from Indian River Lagoon and Banana River, which contain seagrasses). A weight-of-evidence approach employing several analytical techniques was proposed to derive numeric criteria. Approaches included a reference period analysis, chl-a versus concentration of TN or TP regression analyses, and two general models (Dettmann 2001; Steward and Lowe 2010). The general models predicted TN and TP using external loading and water residence time and the reference period method provided analysis to identify desirable trophic state and attainment of TMDL seagrass targets.

The reference condition approach was based on the period from 2000 to 2008. That period was selected because of the low TN levels compared to the previous decade; the low chl-a concentrations during the period that are consistent with chl-a targets established for other estuaries throughout the state; and the good trophic status shown by TSI values of less than 50. Concentrations were calculated using annual median concentrations and maximum wet-season median concentrations (as the highest monthly values from July to September) of TN, TP, and chl-a. Simple linear regressions were used as a second line of evidence to calculate TN and TP criteria on the basis of chl-a targets established by the reference period calculations. The general nutrient models of Steward and Lowe (2010) and Dettmann (2001) were used as a final method with which to estimate loading limits and concentrations associated with those limits. Proposed loading and concentration criteria for the North Halifax River Estuary are based on the loading and concentration estimates of the general nutrient models with estimates of loadings from wastewater treatment facilities (WWTFs) in the estuary removed. The current estimated loadings (as of 2004) of TN and TP and the current concentrations based on the reference approach were proposed by SJRWMD as criteria for the South Halifax River Estuary. Target chl-a values for both segments were calculated using the reference period approach (Table 2-151). The results from the multiple lines of evidence were comparable. Documentation of SJRWMD-proposed numeric nutrient criteria and methods for their derivation can be found in Volume 1, Appendix G.

Table 2-151. SJRWMD-proposed annual median chl-a targets and reference period approach TN and TP concentrations (Steward et al. 2010)

Estuary Segment	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)
North Halifax River Estuary	4.5	0.61	0.12
South Halifax River Estuary	3.5	0.39	0.12

2.19.10. References

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2.20. Guana, Tolomato, Matanzas, Pellicer System

2.20.1. Proposed Numeric Nutrient Criteria

Proposed numeric nutrient criteria for TN, TP, and chl-a in Guana, Tolomato, Matanzas, and Pellicer (GTMP)[90] River Estuary segments are summarized in Table 2-152.

Table 2-152. Proposed numeric nutrient criteria for GTMP River Estuary segments

Segment Name	Segment Number	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
Tolomato River	1701	0.77	0.144	9.5
Matanzas River	1702	0.53	0.108	6.1

[90] The Guana, Tolomato, Matanzas, and Pellicer River Estuary is also called GTM Estuary.

2.20.2. General Characteristics

2.20.2.1. System Description

The GTMP River Estuary is on the northeast coast of Florida near St. Augustine in St. Johns and Flagler counties (Haydt and Frazel 2003). [91] The GTMP Estuary is a long, narrow, marshy, tidal lagoon between mainland Florida and barrier beaches to the east. The GTMP Estuary connects through the Intracoastal Waterway to the St. Johns River Estuary to the north and the Halifax River Estuary to the south and directly to the Atlantic Ocean via two inlets—the St. Augustine Inlet to the north and the Matanzas Inlet to the south (FDEP 2008). The GTMP Estuary is a shallow estuary approximately 50 mi (80 km) long and has an average width of 0.5 mi (0.8 km) in St. Johns County and 0.2 mi (0.3 km) in Flagler County. The GTMP watershed area (395 mi^2 [1,023 km^2]) to estuary area (17 mi^2 [44 km^2]) ratio is approximately 23-to-1, indicative of a potentially large volume of runoff to a proportionally small receiving estuarine system (Steward et al. 2010a).

A large portion of the GTMP Estuary is within the boundaries of the Guana-Tolomato-Matanzas National Estuarine Research Reserve (GTMNERR). The reserve covers an area approximately 101 mi^2 (261 km^2), including salt marsh and tidal wetlands, estuarine lagoons, upland habitat, and offshore seas (FDEP 2009; Frazel 2009).

The GTMP Estuary has undergone significant hydromodification. In 1961 the Guana Dam was built along with a series of smaller canals and dams for improved fishing and hunting. In addition, the Intracoastal Waterway was constructed through the Northern Coastal Basin, connecting many of those estuaries through canals, and requiring continual maintenance with dredging to keep channels clear. Other hydromodifications have included mosquito ditching, dikes, wells, drainage ditches, and land clearing (FDEP 2008; Frazel 2009).

The GTMNERR watershed is characterized by a humid, subtropical, marine climate with heavy rainfall in the summers and mild, dry winters (Frazel 2009). Whereas prevailing winds are easterly, winds from all directions are fairly common. Periodic thunderstorms, northeasters, tropical storms, and hurricanes occur in the GTMP Estuary. In 2004 three tropical storms passed through the area causing extensive beach erosion in St. Augustine (Frazel 2009).

Despite significant population growth, the watershed remains largely undeveloped with around 80 percent of the overall area in forests, wetlands, or surface water in 2000 (SJRWMD 2011b). Urban and residential development is most prevalent in the north Matanzas watershed around St. Augustine (28.1%) (Steward et al. 2010a). The population is centered along the coast and surrounding waterways, with the largest communities bordering the GTMP Estuary (US Census Bureau 2012).

Two ocean inlets, the St. Augustine and Matanzas, and the Intracoastal Waterway connections to the north and south provide tidal flushing and create three tidal nodes in the GTMP Estuary where net flow is zero (Steward et al. 2010a). Both inlets allow substantial tidal exchange, with

[91] The information presented in this system description was compiled to summarize local information pertaining to the proposed numeric nutrient criteria for this estuary. For more information on EPA's process of delineating, segmenting, and deriving numeric nutrient criteria for this estuary, please see Section 1.3.

an average tidal amplitude of around 4.10 ft (1.25 m) (Frazel 2009; Steward et al. 2010a). Closer to the tidal nodes, the tidal amplitude decreases greatly (Haydt and Frazel 2003).

Research in the GTMP Estuary has shown that water residence time is useful in explaining nutrient concentration spatial variations and phytoplankton populations (Phlips et al. 2004; Sheng et al. 2008). Phlips et al. (2004) and Sheng et al. (2008) estimated residence times or residence time indices for each estuary segment. As noted in both studies, tidal flushing is the primary mode of water exchange in the Halifax River Estuary and GTMP Estuary; the flushing time for segments is faster closer to the inlets, and residence time is shorter.

2.20.2.2. Impaired Waters[92]

Nine Class II and Class III marine WBIDs in the GTMP Estuary have been listed for a nutrient-related parameter on Florida's CWA section 303(d) list approved by EPA. Of the nine WBIDs, five are Class II WBIDs, and four are Class III marine WBIDs. Of the five Class II WBIDs, three are impaired for DO (WBIDs 2363F, 2363I, and 2451), one is impaired for nutrients and chl-a (WBID 2320F), and one is impaired for nutrients, chl-a, and DO (WBID 2320). Of the four Class III marine WBIDs, three are impaired for DO (WBIDs 2363H, 2400, and 2491), and one is impaired for nutrients, chl-a, and DO (WBID 2320A).[93] No Class II or Class III marine WBIDs with nutrient-related TMDLs are documented for this region.[94]

The GTMP Estuary was designated a priority water under the SWIM Act. In 2003 a SWIM plan was completed, which created a framework for projects to be completed to reduce point and nonpoint source nutrient contributions (Haydt and Frazel 2003).

2.20.2.3. Water Quality

Nonpoint sources have been estimated to contribute approximately 68 percent of the overall nutrient loading, followed by wastewater treatment dischargers, which contribute 27 percent (Steward et al. 2010a). Septic tanks are also prevalent in some areas of the GTMP Estuary and have been identified as a potential source of nutrients to surface waters (FDEP 2008; SJRWMD 2000b).

Median DO readings at SJRWMD sites at Tolomato River and Pellicer Creek were 6.0 and 4.2 mg/L, respectively, between 1996 and 2011. Median DO levels in the upper and lower Matanzas River are both approximately 6.5 mg/L (SJRWMD 2011a). The extent to which DO concentrations in the GTMP Estuary are influenced by anthropogenic nutrient inputs,

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[92] For more information about the data source, see Volume 1, Appendix A.

[93] The nutrient-related 303(d) list was a compilation of EPA-approved 303(d) listing information for nutrients, chl-a, and DO provided in three decision documents: September 2, 2009, Basin Groups 1, 2, and 5 EPA Decision Document (http://www.epa.gov/region4/water/tmdl/florida/documents/fl09303d_decisiondoc_090209.pdf);
May 13, 2010, Basin Group 3 EPA Decision Document
(http://www.epa.gov/region4/water/tmdl/florida/documents/fl303d_%20partialapproval_decision_docs051410.pdf);
and the December 21, 2010, Basin Group 4 EPA Decision Document
(http://www.epa.gov/region4/water/tmdl/florida/documents/group_4_final_dec_doc_and_partial_app_letter_12_21_10.pdf).

[94] TMDLs were identified in February 2011 by compiling nutrient-related draft/final TMDLs from the following three sources:
FDEP TMDL website (http://www.dep.state.fl.us/water/tmdl/index.htm)
EPA Region 4 website (http://www.epa.gov/region4/water/tmdl/florida/index.html)
EPA National WATERS expert query tool (http://www.epa.gov/waters/tmdl/expert_query.html).

biochemical oxygen demand (BOD) inputs, and import of low-DO water from freshwater streams has not been determined (FDEP 2008).

Phlips et al. (2004) reported mean chl-a concentrations at four sampling locations in the GTMNERR from monthly samples taken May 2002 through August 2003. Overall, chl-a concentrations were lower for GTMP sites than for other sites sampled in estuaries farther north on Florida's Atlantic Coast. Among the four sites studied by Phlips et al. (2004), mean chl-a concentrations were lowest at the San Sebastian and Fort Matanzas sites. Chl-a concentrations at all sites were generally elevated during summer months, associated with increased temperature, increased nutrient loading (wet season), or a combination of the two (Phlips et al. 2004). Median and mean chl-a concentrations measured in North Matanzas, South Matanzas, and Tolomato were low (maximum wet season medians of 4.0–6.4 µg/L), and few samples exceeded 12 µg/L (Steward et al. 2010a).

Steward et al. (2010a) also report turbidity ranges for water quality monitoring locations along the GTMP Estuary from 1986–2009. The Tolomato region had the highest median value (9.4 NTU) compared to the North Matanzas (7.5 NTU) and South Matanzas (7.8 NTU) regions, but the differences were within confidence intervals. Wet-season turbidity was higher than dry-season turbidity at all stations, with seasonal differences in the 1–4 NTU range. The highest turbidities (both wet and dry season) occurred at sampling locations with little or no tidal influence (the southernmost sampling point in the Intracoastal Waterway). High turbidity throughout the GTMP Estuary is consistent with other estuaries along the south Atlantic Coast with limited tidal connections (Steward et al. 2010a).

There were statistically significant increasing trends in TP and chl-a concentrations over a 23-year period (1986–2009) in the GTMP Estuary. The early portion of the time series coincided with several drought years, during which concentrations would have been lower because of reduced nutrient inputs. Long-term median and mean TP concentrations were 0.072–0.084 and 0.087–0.10 mg/L, respectively. For the same period, TN concentrations were similar throughout the estuary, with the exception of the southernmost station where the TN concentrations were significantly higher. The long-term median and mean TN concentrations in the GTMP Estuary were 0.41–0.53 and 0.43–0.58 mg/L, respectively (Steward et al. 2010a).

2.20.2.4. Biological Characteristics

The GTMP Estuary is primarily characterized by tidal salt marshes and oyster reefs (Dame et al. 2000; Sargent et al. 1995; Steward et al. 2010a). Wetlands compose about one-third of the overall area of the GTMP Estuary watershed (Steward et al. 2010a). Salt marshes are the predominant wetland community and make up about 20 percent of the overall land cover of the GTMNERR (Frazel 2009). SAV is largely absent, as it is in much of northeastern coastal Florida, from lack of suitable habitat and elevated turbidity (Dame et al. 2000; Sargent et al. 1995; Steward et al. 2010a).

Documented impacts on shoreline vegetation in the GTMP Estuary have been linked to factors other than anthropogenic nutrient loading. Shoreline erosion and resultant loss of shoreline habitat occurs at relatively high rates in many places along the Intracoastal Waterway, which runs through GTMNERR (Price 2005).

Oyster populations occur along the full length of the GTMNERR. As of 2009, two delineated shellfish harvest areas in the GTMNERR allowed limited recreational oyster and hard clam harvesting. In addition, four active aquaculture leases for oysters and two leases for hard clams exist (Frazel 2009). Inclusion in the GTMNERR and other reserves has likely helped preserve these oyster reefs (Steward et al. 2010b).

Phytoplankton abundance in the GTMP is primarily regulated by a balance between water residence time and nutrient loading, with water residence time determined by multiple factors including proximity to inlets, freshwater inputs, and vertical mixing (Phlips et al. 2004). In the 2002–2003 study conducted by Phlips et al. (2004), the authors attribute low chl-a concentrations (as a surrogate measure for phytoplankton crop) in the vicinity of the GTMP Estuary to its close proximity to tidal inlets and lower residence time.

Frazel (2009) reported that little research has been done on plankton communities in the GTMNERR, but that there are periodic blooms of *K. brevis* (a ride tide organism) on Florida's east coast. A *K. brevis* bloom occurred in GTMNERR waters in October 2007 (Frazel 2009). Abbott et al. (2009) classified the GTMP Estuary and associated coastal waters as affected by toxins associated with HABs, including neurotoxic shellfish poisoning. Blooms can arise from either chronic or episodic nutrient loading (Heisler et al. 2008).

A study conducted for the SJRWMD in 2000 included five benthic macroinvertebrate sampling sites in the GTMP Estuary. While the coverage was limited, all five sites had moderate to high species diversity scores (Shannon-Weiner Diversity Index).[95] The three southern sites exhibited species compositions that were dominated by pollution-tolerant species, indicating the possible presence of pollutants in Moultrie Creek, Pellicer Creek, and the South Matanzas River. The two sites in the northern areas on the North Matanzas River and the Tolomato River had low-tolerant taxa dominance (SJRWMD 2000a).

Research has identified 303 fish species in the GTMNERR. A number of commercially important species are in the estuary, as are many recreationally valuable sport fish (Frazel 2009). Twenty-five accounts of fish kills resulting from algae or red tide were reported from 2000 to 2010 in Flagler and St. Johns counties, some of which were not in the GTMP Estuary. Overall, 68 percent of those reports were the result of a red tide event in 2007 that was documented to have drifted into the GTMP Estuary from further north up the Atlantic Coast (FFWCC 2011; Frazel 2009).

For a more detailed summary of this water body, see Volume 1, Appendix A.

[95] Shannon-Weiner Species Diversity Index is "a calculated index value expressing the degree of species diversity in a given sample or group of samples. The calculation is influenced by both the number of species present as well as the evenness of abundance among the species. Values generally range from 0 to 5, with values at the high range indicating high species diversity" (SJRWMD 2000a).

2.20.3. Data Used

Several data sources specific to the GTMP System were used in addition to those sources described in Section 1.4.3, as summarized in Table 2-153.

Table 2-153. Data sources specific to GTMP System models

Data	Source	Location Used
Hydrologic group soils data	St. Johns River Water Management District (SJRWMD 2011c)	Daytona watershed model
FDEP Level III Florida Land Use	St. Johns River Water Management District (SJRWMD 2006)	Daytona watershed model

2.20.4. Segmentation

The GTMP system was divided on the basis of its geomorphological structure and the different river systems feeding into it. Figure 2-57 shows the resulting two segments for the GTMP Estuary system.

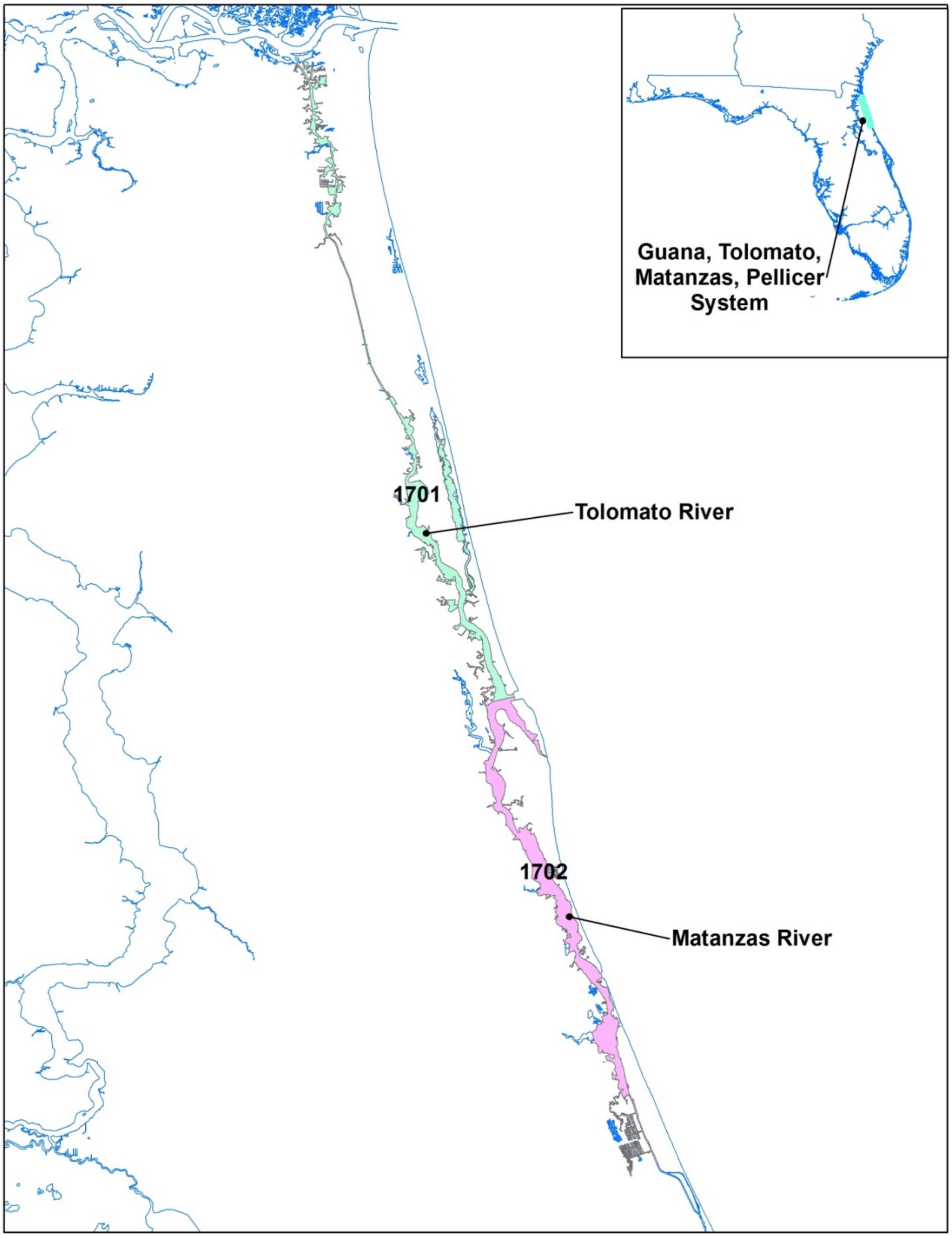

Figure 2-57. Results of GTMP Estuary segmentation

2.20.5. Water Quality Targets

2.20.5.1. Seagrass Depth and Water Clarity Targets

Seagrass is not historically present in the GTMP Estuary.

2.20.5.2. Chlorophyll a Target

To prevent nuisance algal blooms and protect the estuary's designated uses, chl-a levels must not exceed 20 µg/L more than 10 percent of the time. The rationale for that target is provided in Section 1.2.2.

2.20.5.3. Dissolved Oxygen Targets

On the basis of the rationale that sufficient DO is necessary to protect aquatic life, as described in Section 1.2.3, the following DO targets were established:

- Minimum allowable DO of 4.0 mg/L as a water column average for each estuary segment 90 percent of the time over the simulation's time span
- Daily average DO of 5.0 mg/L as a water column average for each estuary segment 90 percent of the time over the simulation's time span
- Minimum 3-hour average DO of 1.5 mg/L in the bottom two layers for each estuary segment over the simulation's time span

2.20.6. Results of Analyses

2.20.6.1. Mechanistic Model Analysis

The GTMP system was not evaluated using mechanistic model analysis.

2.20.6.2. Statistical Model Analysis

Analysis of available empirical data indicated a strong relationship between the bloom frequency endpoint and annual geometric mean chl-a concentration (Figure 2-58). In both segments, the derived chl-a concentration was greater than the upper bound of the observed annual geometric mean, so chl-a criteria are based on this upper bound.

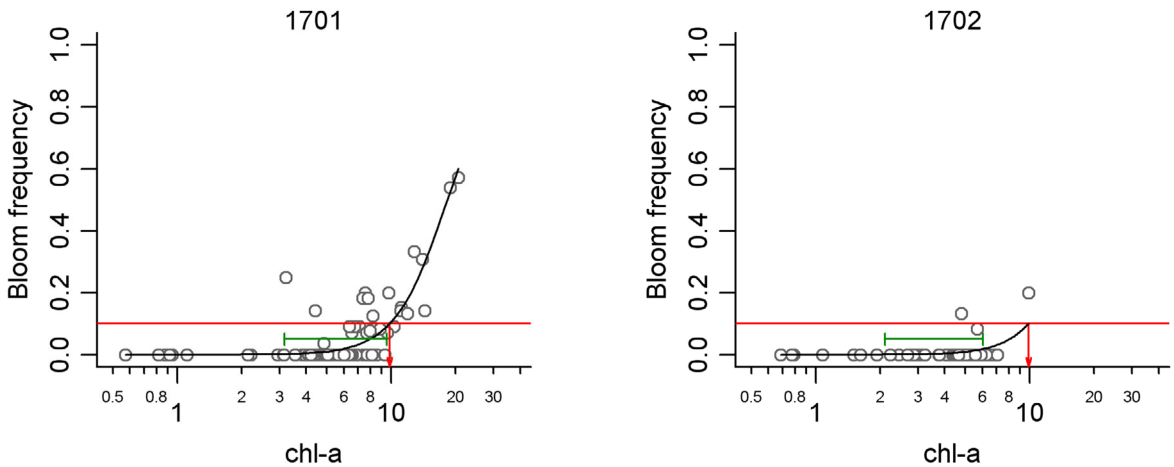

Figure 2-58. Estimates of annual geometric chl-a concentrations associated with bloom frequency of 0.1 in GTMP. Red horizontal line: bloom frequency of 0.1, red vertical arrow: annual geometric mean chl-a concentration associated with 0.1 bloom frequency, green line segment: 5th to 95th percentile range of observed data.

In general, chl-a concentrations increased with increasing concentrations of TN and TP. However, in segment 1702, a negative correlation between TN and chl-a was observed, suggesting that TN in this segment is primarily composed of recalcitrant forms (Figure 2-59). TN and TP concentrations that were associated with the candidate chl-a criterion were all greater than the upper bound of observed annual geometric means, and therefore, proposed criteria were based on the upper bound (Table 2-154). A TN criterion in segment 1702 was based on the dilution model described below.

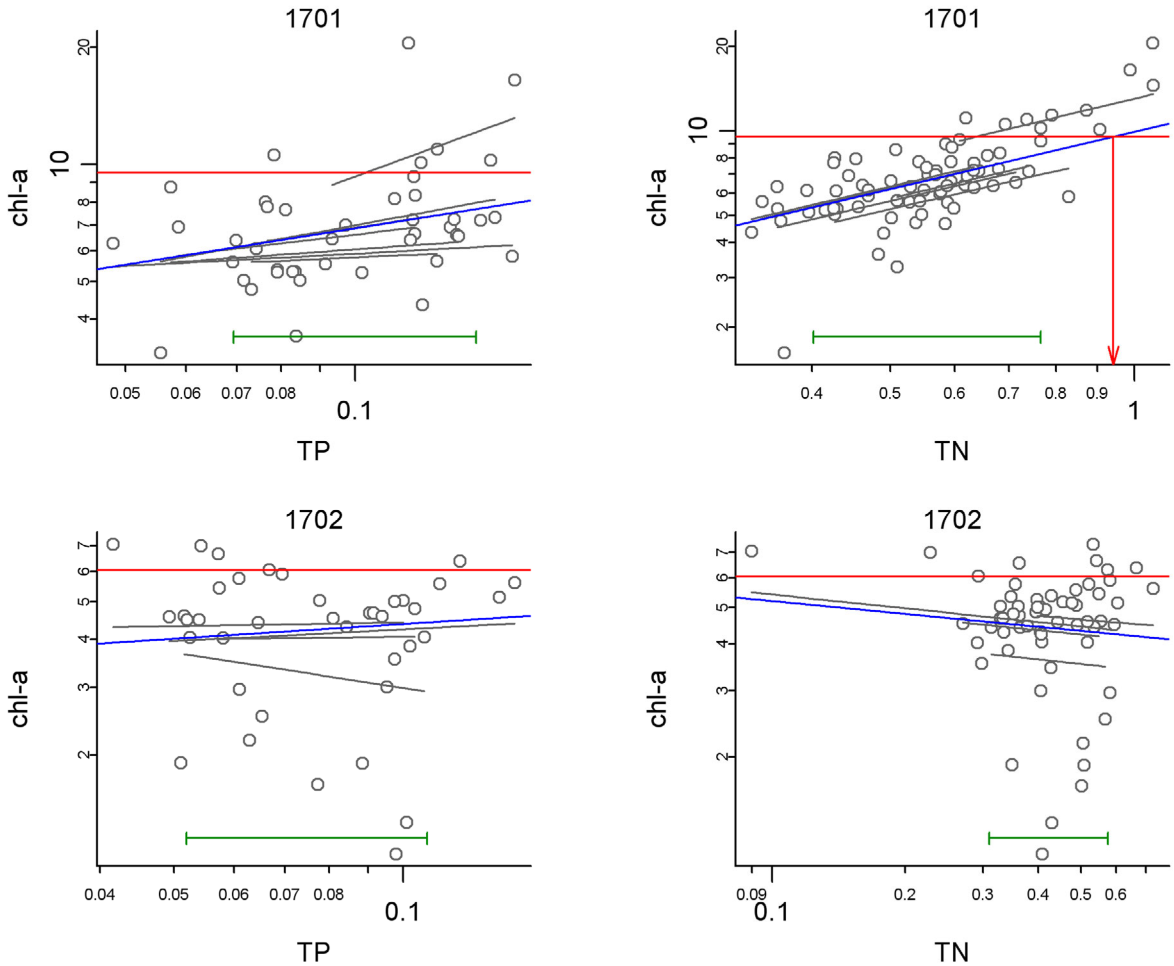

Figure 2-59. Relationships between TN, TP, and chl-a in GTMP. Open circles: observed annual average values of TN, TP, and chl-a, red horizontal line: chl-a criterion, red vertical arrow: TN and TP criteria associated with chl-a criterion, blue line: estimated segment-wide relationship between TN, TP, and chl-a, grey lines: estimated station-specific relationships between TN, TP, and chl-a, green line segment: 5th to 95th percentile range of observed annual geometric mean TN and TP concentrations.

Table 2-154. Summary of candidate criteria for GTMP. TN criterion for segment 1702 is based on dilution model. Asterisks indicate that criteria are based on the upper or lower bound of observed data.

Segment	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
1701	0.77*	0.144*	9.5*
1702	0.53	0.108*	6.1*

2.20.7. Application of Analyses for Proposed Numeric Nutrient Criteria

In GTMP data were sufficient to use statistical modeling analyses as the primary line of evidence when deriving criteria. Seagrass has not been historically present in GTMP, so EPA evaluated the following endpoint in the statistical modeling approach: chl-a concentrations associated with balanced phytoplankton biomass. Through evaluating the observed data, EPA found that, in some segments, the TN, TP, and chl-a concentrations associated with achieving the chl-a target were greater than the range of TN, TP, or chl-a concentrations observed in the available data for GTMP. For these segments, EPA is proposing to set numeric nutrient criteria derived from statistically modeled relationships at the upper bound of the distribution of available data instead of deriving criteria outside the range of data observations (see Volume 1, Appendix B). This approach defines criteria values that maintain balanced natural populations of aquatic flora and fauna within the limits of available data.

Proposed numeric nutrient criteria for TN, TP, and chl-a in GTMP segments are summarized in Table 2-155.

Table 2-155. Proposed and candidate numeric nutrient criteria for Guana, Tolomato, Matanzas, Pellicer segments

		Proposed Criteria			Statistical Modeling		
SEGMENT	SEGMENT ID	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
Tolomato River	1701	0.77	0.144	9.5	0.77	0.144	9.5
Matanzas River	1702	0.53	0.108	6.1	0.53	0.108	6.1

2.20.8. Downstream Protective Values

In GTMP mechanistic models were not available to derive DPVs for TN and TP. In lieu of the preferred approach, a mixing/dilution model was applied to calculate the allowable freshwater TN and TP load. This mixing/dilution model assumes that TN and TP loads in freshwater mix conservatively with saline seawater. Using this assumption, the model predicts freshwater concentrations necessary to achieve proposed nutrient criteria in each segment (Figure 2-60 and Figure 2-61). DPVs are tabulated in Table 2-156.

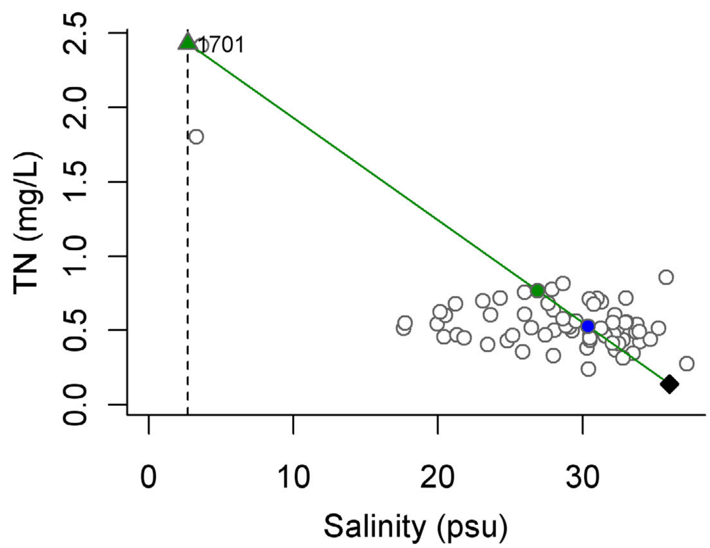

Figure 2-60. Calculation of TN DPVs for GTMP. Black diamond shows seawater conditions, filled green circles show proposed TN criteria values for each segment, and green triangles show calculated DPVs. Open circles show observed long-term station average TN concentrations and salinities. Filled blue circle shows TN criteria computed for segment 1702.

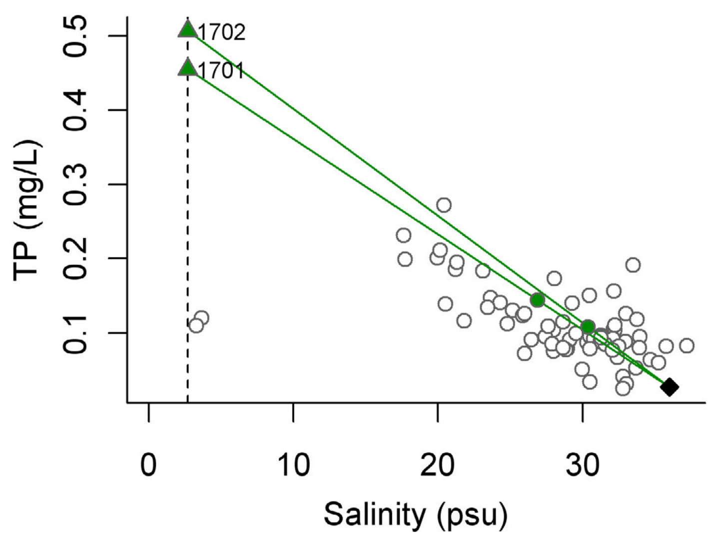

Figure 2-61. Calculation of TP DPVs for GTMP. Black diamond shows seawater conditions, filled green circles show proposed TP criteria values for each segment, and green triangles show calculated DPVs. Open circles show observed long-term station average TP concentrations and salinities.

Table 2-156. Proposed DPVs for TN and TP for GTMP

Segment	DPV (TP) (mg/L)	DPV (TN) (mg/L)
1701	0.455	2.43
1702	0.507	2.43

2.20.9. Alternate Analysis

2.20.9.1. Tolomato–Matanzas Estuary

SJRWMD submitted documents to EPA suggesting approaches to derive numeric criteria for the Tolomato–Matanzas Estuary (TME). A weight-of-evidence approach using several analytical techniques was proposed to derive numeric criteria (Table 2-157). The techniques included a reference period analysis, chl-a versus concentration of TN or TP regression analyses, and two general models (Dettmann 2001; Steward and Lowe 2010).

TN and TP loading, chl-a target concentrations, and TN and TP concentration criteria were based on an approach that analyzed water quality and estimated loading during a reference period from 2000 to 2009. The period of reference was selected on the basis of a desirable TSI score (< 50), rainfall amounts typical of average conditions, and completeness of the data record. Criteria magnitudes were proposed as an annual median or mean and a maximum wet-season (June–September) median or mean. The reference period approach results were supported by an additional line of evidence using regression analyses of chl-a versus TN and TP. Target chl-a values were based on the reference period analyses. The general nutrient models of Steward and Lowe (2010) and Dettmann (2001) were also used as an additional method by which to estimate loading limits and concentrations associated with those limits. Documentation of SJRWMD-proposed numeric nutrient criteria and methods for their derivation can be found in Volume 1, Appendix G.

Table 2-157. SJRWMD-proposed TME loading limits for TN, TP, and chl-a according to reference period results (adapted from Steward et al. 2010a)

Estuary Segment	Chl-a (µg/L) median (mean)	TN (mg/L) median (mean)	TP (mg/L) median (mean)
Tolomato	4.5 (5.3)	0.52 (0.56)	0.085 (0.096)
North Matanzas	3.1 (3.5)	0.37 (0.41)	0.073 (0.083)
South Matanzas	4.3 (5.0)	0.45 (0.49)	0.089 (0.103)

2.20.10. References

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2.21. St. Johns River

2.21.1. Proposed Numeric Nutrient Criteria

Proposed numeric nutrient criteria for TN, TP, and chl-a in St. Johns River segments are summarized in Table 2-158.

Table 2-158. Proposed numeric nutrient criteria for St. Johns River segments

Segment Name	Segment Number	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
Lower St. Johns River	1801	0.75	0.095	2.5
Middle St. Johns River	1802	1.09	0.108	3.6
Upper St. Johns River	1803	1.15	0.074	7.7

2.21.2. General Characteristics

2.21.2.1. System Description

The St. Johns River is a broad, shallow, slow-moving, blackwater river that flows northward about 300 mi (480 km), making it the longest river in Florida (FDEP 2002, 2011b; UNF and JU 2009). The greater watershed drains 9,562 mi^2 (24,765 km^2) of northeast Florida, nearly one-fifth of the state's total surface area (Hendrickson and Konwinski 1998). The Lower St. Johns River is a dark, blackwater river. Southern blackwater rivers are colored by high loads of suspended matter and the release of organic acids from decomposing forest leaf litter. Color limits light penetration to a shallow layer near the surface, which shapes plant communities throughout blackwater rivers (Brody 1994; Phlips et al. 2000).

The watershed is commonly divided into the Upper St. Johns River Basin, Middle St. Johns River Basin, Lake George Basin, and Lower St. Johns River Basin.[96] The focus of this summary is the Lower St. Johns River Basin, which is the tidally influenced portion of the St. Johns River between the mouth of the Ocklawaha River and the Atlantic Ocean, 100 mi (161 km) downstream (FDEP 2008; Phlips et al. 2000; SJRWMD 2008). The Lower St. Johns River drains approximately 2,750 mi^2 (7,122 km^2) of northeast Florida (FDEP 2008). Slow flushing is a concern in the estuary (Phlips et al. 2000). The hydrology of the Lower St. Johns River Basin is influenced by several factors, including the tide, wind, freshwater flows, and channel restrictions (FDEP 2011a).

The Lower St. Johns River Basin is characterized as warm temperate to subtropical, with an annual average temperature of 69.8 °F (21 °C) and annual average rainfall of 53 in (135 cm; climate records from 1971 to 2000) (FDEP 2002). The Lower St. Johns River Basin experiences a predictable pattern of wet and dry seasons, is prone to storms (e.g., hurricanes and tropical storms), and has experienced droughts that affect the characteristics of the river (FDEP 2002; UNF and JU 2009).

[96] The information presented in this system description was compiled to summarize local information pertaining to the proposed numeric nutrient criteria for this estuary. For more information on EPA's process of delineating, segmenting, and deriving numeric nutrient criteria for this estuary, please see Section 1.3.

The Lower St. Johns River Basin has experienced high population growth, and this trend is expected to continue. Flagler, Clay, and St. Johns counties have experienced the greatest population growth in the basin. Duval County, which includes Jacksonville, is the most urban, densely populated area in the basin (FDEP 2002; SJRWMD 2008). Heavy mineral mining occurs at the northwestern boundary of the basin, and sand and gravel are mined in Clay and Putnam counties. Silvicultural operations are widespread throughout the basin, and agriculture is mostly concentrated in Flagler, St. Johns, and Putnam counties (Tri-County Agricultural Area). The Timucuan Ecological and Historic Preserve is undeveloped and covers approximately 72 mi^2 (186 km^2 [46,000 ac]) (FDEP 2002, 2011a; SJRWMD 2008). Land use in the greater St. Johns River watershed is predominantly forested (28%), followed by wetlands (25%), agricultural lands (18%), and urban areas (16%) (Fry et al. 2011; SJRWMD 2006; SRWMD No date; SWFWMD 2007).[97]

2.21.2.2. Impaired Waters[98]

Four Class III marine WBIDs in the Lower St. Johns River Basin are currently listed for a nutrient-related parameter on Florida's 2010 CWA section 303(d) list approved by EPA. Of the four Class III marine WBIDs, two are impaired for nutrients and chl-a (WBIDs 2228 and 2265A); one is impaired for nutrients, chl-a, and DO (WBID 2191); and one is impaired for nutrients (WBID 2205C). There are no Class II WBIDs with nutrient-related impairments documented for this area.[99]

Three nutrient-related TMDLs for Class II or Class III marine WBIDs exist in the Lower St. Johns River Basin. One of the TMDLs is the final *Lower St. Johns River Nutrients TMDL,* covering Class III marine WBIDs (2213A, 2213B, 2213C, 2213E, 2213F). The other two are the final *Moncrief Creek Nutrient TMDL,* covering Class III marine WBID 2228, and the final *Arlington River Nutrient TMDL*, covering Class III marine WBID 2265A; both of those WBIDs are currently listed on Florida's 2010 CWA section 303(d) list approved by EPA.[100]

2.21.2.3. Water Quality

The Lower St. Johns River Basin has a long history of hydromodification, with wetlands lost to agriculture, development, and clearing of forests for timber. Population growth in the Lower St. Johns River Basin continues to put pressure on the natural landscapes (FDEP 2002).

[97] See Volume 1, Appendix C and its attachments for a summary of how land use data was combined and for a detailed breakdown of land uses.

[98] For more information about the data source, see Volume 1, Appendix A.

[99] The nutrient-related 303(d) list was a compilation of EPA-approved 303(d) listing information for nutrients, chl-a, and DO provided in three decision documents: September 2, 2009, Basin Groups 1, 2, and 5 EPA Decision Document (http://www.epa.gov/region4/water/tmdl/florida/documents/fl09303d_decisiondoc_090209.pdf);
May 13, 2010, Basin Group 3 EPA Decision Document
(http://www.epa.gov/region4/water/tmdl/florida/documents/fl303d_%20partialapproval_decision_docs051410.pdf);
and the December 21, 2010, Basin Group 4 EPA Decision Document
(http://www.epa.gov/region4/water/tmdl/florida/documents/group_4_final_dec_doc_and_partial_app_letter_12_21_10.pdf).

[100] TMDLs were identified in February 2011 by compiling nutrient-related draft/final TMDLs from the following three sources:
FDEP TMDL website (http://www.dep.state.fl.us/water/tmdl/index.htm)
EPA Region 4 website (http://www.epa.gov/region4/water/tmdl/florida/index.html)
EPA National WATERS expert query tool (http://www.epa.gov/waters/tmdl/expert_query.html).

Nonpoint sources, such as agricultural and urban runoff, and point sources, such as domestic and industrial WWTFs, have been identified as significant sources of nutrients, bacteria, and toxic pollution to the Lower St. Johns River Basin (SJRWMD 2008). In addition, a study in Duval County identified shallow groundwater as a likely transport mechanism of nutrients and fecal bacteria from septic tank leachate to surface waters (Wicklein 2004).

No clear trend in annual DO water quality from the STORET database in the Lower St. Johns River and Estuary was apparent from 1967 to 2007. However, a seasonal analysis revealed that DO levels in the Lower St. Johns River tend to be lower during the summer, especially in the tributaries to the Lower St. Johns River. The average June DO concentration in the Lower St. Johns River and Estuary between 1967 and 2007 was 4.5 mg/L (UNF and JU 2009).

Spikes in maximum chl-a measurements are typical of a system with algal blooms (UNF and JU 2009). For further information on chl-a values, see Volume 1, Appendix A. For more information about algal blooms in the Lower St. Johns River, see the biological characteristics section below.

Turbidity data from STORET and Florida's CWA section 303(d) reports from 1970 to 2007 show improvement each decade (i.e., a decrease in turbidity) (UNF and JU 2009).

Average annual TN levels were relatively stable from 1981 to 2007, but data were highly variable (UNF and JU 2009).

Annual average TP concentrations in the Lower St. Johns River and Estuary were generally higher in the early 1970s, followed by a short decline in the late 1970s and an increase in the 1980s. Average TP concentrations gradually decreased and became stable from around 1992 to the 2007 period of record (UNF and JU 2009).

2.21.2.4. Biological Characteristics

The Lower St. Johns River and estuary support a large and diverse community of plant and animal species (FDEP 2002). The Lower St. Johns River is primarily a phytoplankton-based system as a result of limited light (FDEP 2002, 2010a; Sagan 2007).

Wetlands cover about a quarter of the total land area in the Lower St. Johns River Basin. Both freshwater and saltwater wetland communities exist in the Lower St. Johns River Estuary (FDEP 2002). Throughout the 1900s, wetlands were drained with canals, ditches, and levees and were converted to agricultural and urban land (Sparkman 2011). One report noted a trend in the conversion of wetland types from forested wetlands to nonforested wetlands (UNF and JU 2009). In 1973 forested wetlands in the Lower St. Johns River Basin composed 91 percent of total wetlands, but by 2004 only 75 percent of total wetlands were forested (UNF and JU 2009). No further information was found in the available literature.

Healthy blackwater river systems usually have low levels of phytoplankton because of the low light conditions. However, high levels of nutrient pollution in the Lower St. Johns River have led to frequent blooms of blue-green algae (cyanobacteria) and other algae. The Lower St. Johns River has experienced periodic algal blooms of cyanobacteria since the 1800s. However, over the past few decades, algal blooms have increased in frequency (FDEP 2002; UNF and JU 2009). Cyanobacteria can tolerate lower light levels than most other plant species and often outcompete the plants in disturbed blackwater streams (Phlips et al. 2000; UNF and JU 2009). Potentially toxigenic cyanobacteria, such as *Microcystins* and *cylindrospermopsin*, have been reported in large numbers in the St. Johns River, including sites in the Lower St. Johns River, and toxic cyanobacteria such as *Anabaena circinalis* and *Cylindrospermopsis raciborskii* have been implicated in fish kills in the Lower St. Johns River and Estuary (FWRI 2009).

Long-term trends in macroinvertebrate populations were studied and summarized as part of the 2009 *State of the River Report* for the Lower St. Johns River. Substantial changes in the community structure of macroinvertebrates in the northern and southern Lower St. Johns River were recorded between the 1970s and 2000s, highlighted by a shift toward species known to be pollutant-tolerant in both the north and the south during the 2000s (UNF and JU 2009).

The Lower St. Johns River and Estuary is an important commercial and recreational system and serves as nursery grounds for approximately 170 estuarine/marine and freshwater species. Blue crabs compose 76 percent of the overall commercial landings in the Lower St. Johns River, and finfish account for the remaining 24 percent (UNF and JU 2009). Reports of fish kills and external abnormalities are common in the Lower St. Johns River. Local anglers in the late-1980s began reporting large numbers of fish with external lesions called ulcerative disease syndrome (FDEP 2002; Patterson 2010). In the five counties that are part of the Lower St. Johns River, 266 observations of fish kills were reported since 2000 as a result of HABs, and 116 observations were reported of fish with external lesions since 1991. Many of those observations are for multiple fish, sometimes thousands (FFWCC 2011). Florida FWRI data (2001 through 2011) on external abnormalities in fish show a decreasing trend in the frequency of lesions (UNF and JU 2009).

For a more detailed description of this water body, see Volume 1, Appendix A.

2.21.3. Data Used

Several data sources specific to the St. Johns River were used in addition to those sources described in Section 1.4.3, as summarized in Table 2-159.

Table 2-159. Data sources specific to St. Johns River models

Data	Source	Location Used
Hydrologic group soils data	St. Johns River Water Management District (SJRWMD 2011)	St. Johns watershed model
FDEP Level III Florida Land Use	St. Johns River Water Management District (SJRWMD 2006)	St. Johns watershed model
Springs discharge and water quality data	Florida Department of Environmental Protection (FDEP 2004, 2009)	St. Johns watershed model

2.21.4. Segmentation

The GIS isohaline analysis yielded three segments for the St. Johns Estuary. EPA ensured that the segmentation only included waters classified as marine waters (Class III marine). The segmentation scheme is similar to the one proposed by FDEP (FDEP 2010b) for developing numeric nutrient criteria. Figure 2-62 shows the resulting three segments for St. Johns Estuary.

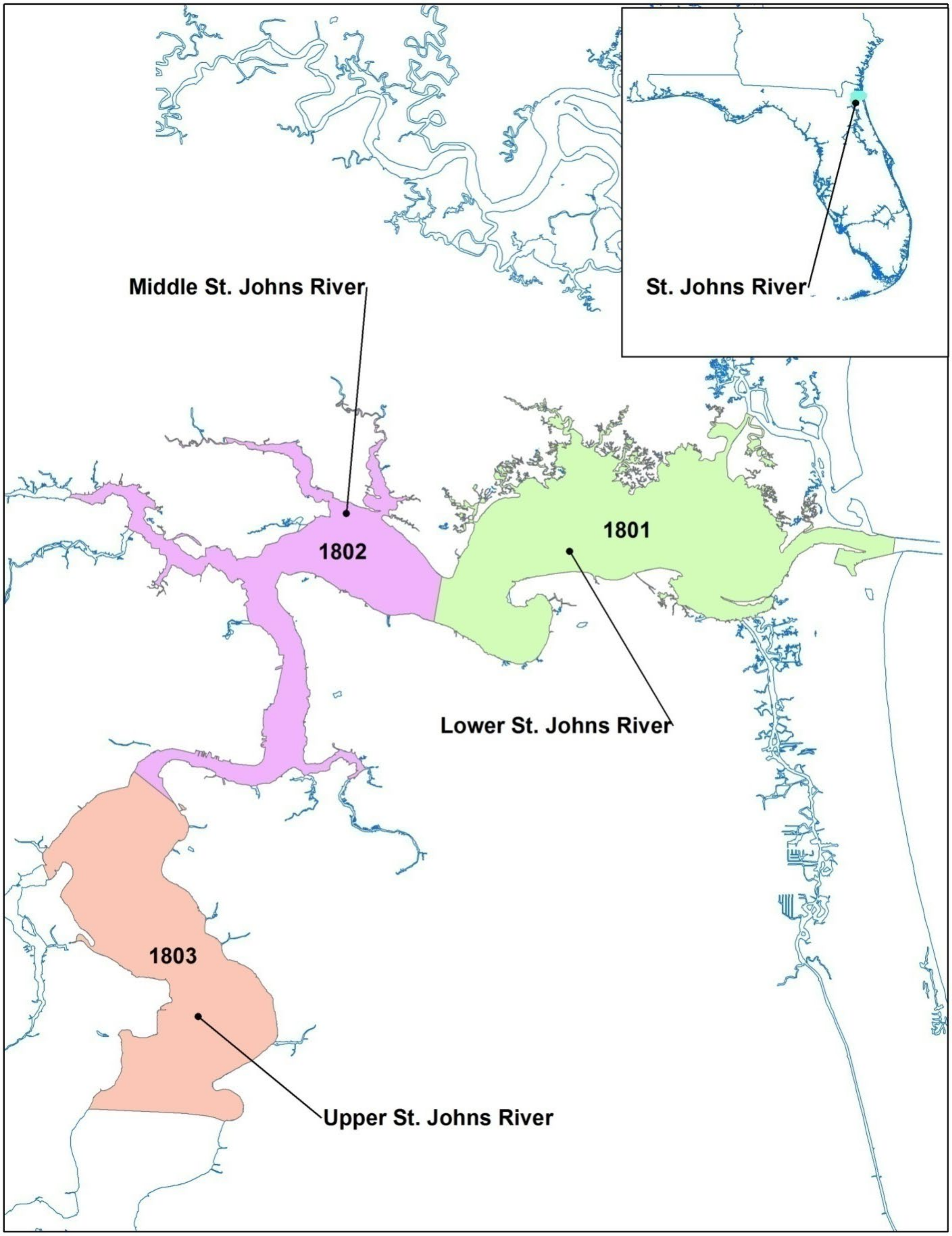

Figure 2-62. Results of St. Johns Estuary segmentation

2.21.5. Water Quality Targets

2.21.5.1. Seagrass Depth and Water Clarity Targets

Seagrass is not historically present in the St. Johns Estuary.

2.21.5.2. Chlorophyll a *Target*

To prevent nuisance algal blooms and protect the estuary's designated uses, chl-a levels must not exceed 20 µg/L more than 10 percent of the time. The rationale for that target is provided in Section 1.2.2.

2.21.5.3. Dissolved Oxygen Targets

Prior to EPA's derivation of numeric nutrient criteria, FDEP developed and adopted a Site-Specific Alternative Criterion (SSAC) for DO for the marine portion of the St. Johns River between Julington Creek and the mouth. This was documented in the publication *Site Specific Alternative Dissolved Oxygen Criterion to Protect Aquatic Life in the Marine Portions of the Lower St. Johns River Technical Support Document* (FDEP and SJRWMD 2006). The SSAC was approved by EPA on October 10, 2006. This DO SSAC was the target for the estuarine portion of the St. Johns River in FDEP's June 2008 *Total Maximum Daily Load for Nutrients for the Lower St. Johns River*.

The SSAC is expressed as follows: The first part of the proposed SSAC is a minimum DO concentration of 4.0 mg/L. In addition, the Total Fractional Exposure to DO levels in the 4.0 to 5.0 mg/L range must also be at or below 1.0 for each annual evaluation period as determined by the equation:

$$\left(\begin{array}{c}\textit{Total Fractional}\\ \textit{Exposure}\end{array}\right) = \frac{\begin{array}{c}\textit{Days between}\\ 4.0\text{–}<4.2\,mg/L\end{array}}{16\ \textit{day max}} + \frac{\begin{array}{c}\textit{Days between}\\ 4.2\text{–}<4.4\,mg/L\end{array}}{21\ \textit{day max}} + \frac{\begin{array}{c}\textit{Days between}\\ 4.4\text{–}<4.6\,mg/L\end{array}}{30\ \textit{day max}} + \frac{\begin{array}{c}\textit{Days between}\\ 4.6\text{–}<4.8\,mg/L\end{array}}{47\ \textit{day max}} + \frac{\begin{array}{c}\textit{Days between}\\ 4.8\text{–}<5.0\,mg/L\end{array}}{55\ \textit{day max}}$$

Where the "Days between..." is the number of days within each interval based on the daily average DO concentration.

Because FDEP developed the DO SSAC and EPA approved the SSAC as a more appropriate criterion to protect aquatic life for this system than the daily average of 5.0 mg/L, EPA used it to derive numeric nutrient criteria. To be consistent with the other estuarine systems, EPA also used DO targets of 4.0 mg/L as a minimum water column average 90 percent of the time and 1.5 mg/L in the bottom two layers as a minimum 3-hour average. These targets were established based on the rationale that sufficient DO is necessary to protect aquatic life, as described in Section 1.2.3.

2.21.6. Results of Analyses

2.21.6.1. Mechanistic Model Analysis

An interconnected suite of basinwide hydrologic, hydrodynamic, and water quality models was assembled to develop the Lower St. Johns River TMDL (FDEP 2008). Those models were run for the TMDL time period of analysis using the DO targets to determine candidate criteria for the

St. Johns River. See Section 5 of the Lower St. Johns River TMDL (FDEP 2008) for a description of the model setup and calibration procedures.

Candidate criteria for St. Johns Estuary segments are given in Table 2-160.

Table 2-160. Summary of candidate criteria for St. Johns derived from mechanistic modeling

Segment	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
1801	0.75	0.095	2.5
1802	1.09	0.108	3.6
1803	1.15	0.074	7.7

2.21.6.2. Statistical Model Analysis

Data were not sufficient in St. Johns River to conduct statistical analysis of the DO endpoint. No water clarity endpoints were available for the St. Johns River, but analysis of chl-a concentrations with respect to the phytoplankton bloom endpoint yielded the following candidate chl-a criteria: 6.1 µg/L, 8.5 µg/L, and 8.4 µg/L for segments 1801, 1802, and 1803, respectively (Figure 2-63).

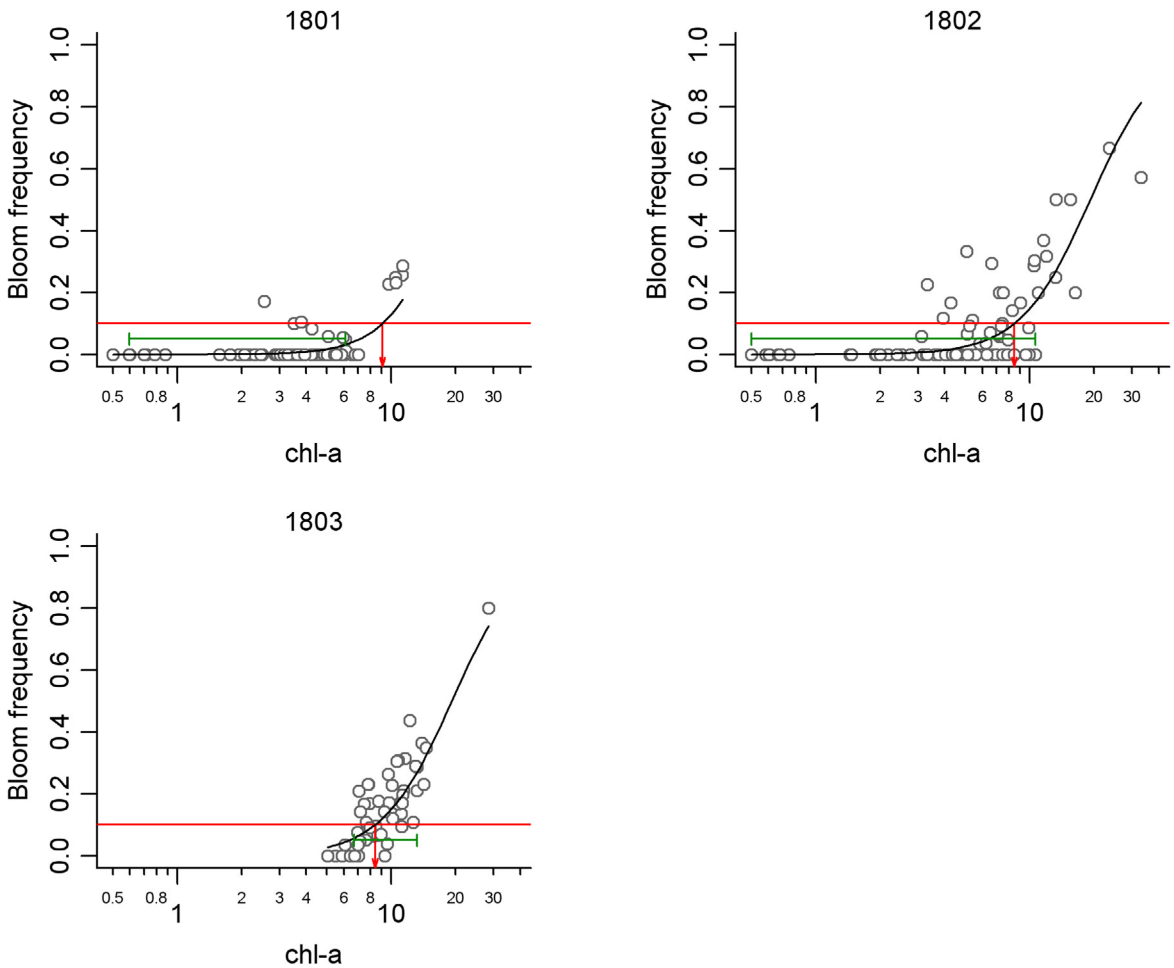

Figure 2-63. Estimates of annual geometric chl-a concentrations associated with bloom frequency of 0.1 in the St. Johns. Red horizontal line: bloom frequency of 0.1, red vertical arrow: annual geometric mean chl-a concentration associated with 0.1 bloom frequency, green line segment: 5th to 95th percentile range of observed data.

Statistical relationships between TN, TP, and chl-a were estimated in the St. Johns using available data (Figure 2-64). Relationships between TN and TP were associated with decreasing concentrations of chl-a in segments 1801 and 1802, likely a consequence of strong influences of upstream conditions, and possibly because TN and TP are dominated by forms that are not biologically available. In segment 1803, the expected increasing relationships between TN, TP, and chl-a were observed, and candidate criteria values were computed. Candidate criteria values for segment 1803 were 0.96 mg/L for TN and 0.11 mg/L for TP.

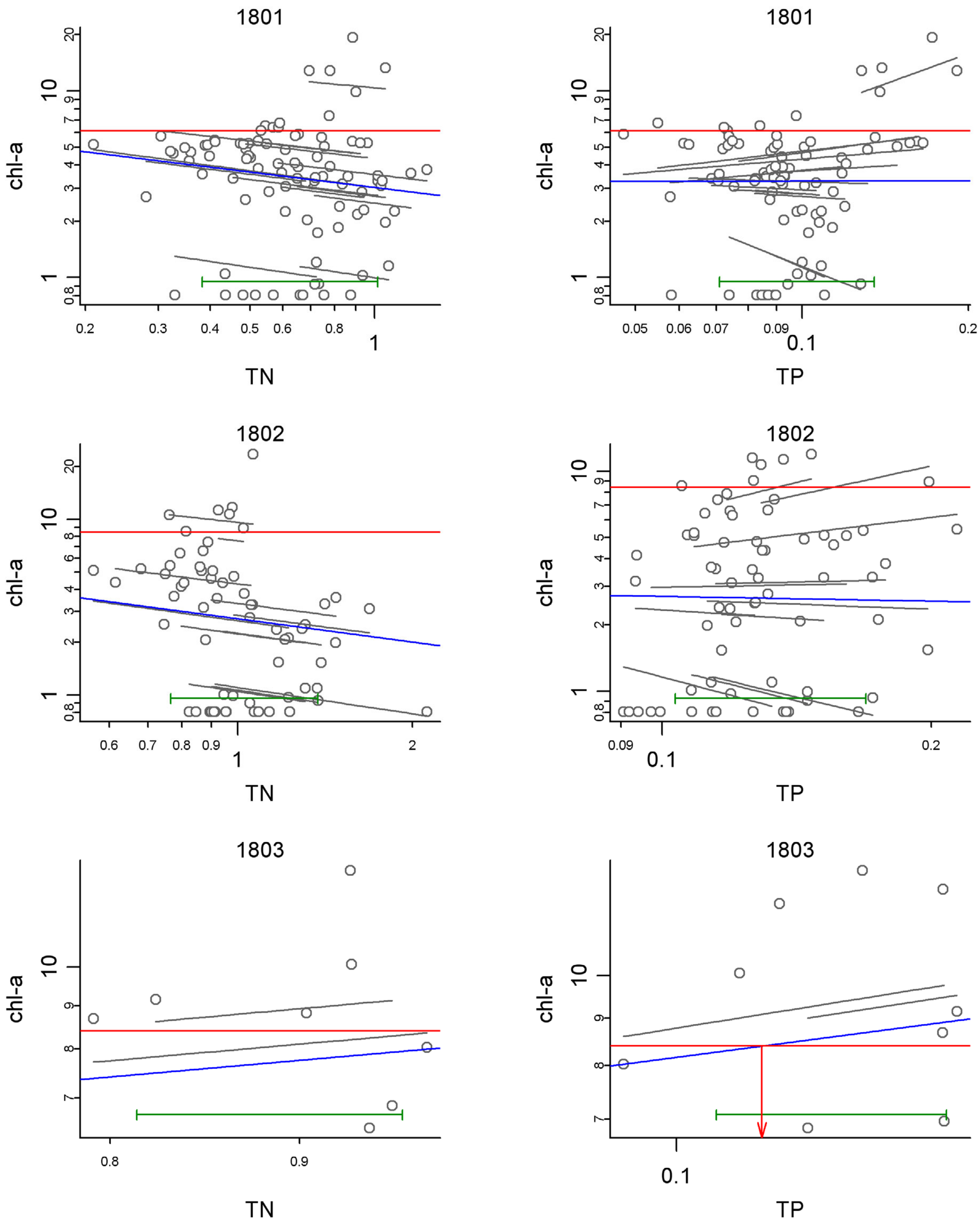

Figure 2-64. Relationships between TP, TN, and chl-a in St Johns. Open circles: observed annual average values of TP, TN, and chl-a, red horizontal line: chl-a criterion, red vertical arrow: TP and TN criteria associated with chl-a criterion, blue line: estimated segment-wide relationship between TP, TN, and chl-a, grey lines: estimated station-specific relationships between TP, TN, and chl-a, green line segment: 5th to 95th percentile range of observed annual geometric mean TP and TN concentrations.

2.21.7. Application of Analyses for Proposed Numeric Nutrient Criteria

EPA evaluated various lines of evidence for deriving criteria in the St. Johns River estuary. Data necessary to conduct statistical analyses were not sufficient. EPA derived the proposed numeric nutrient criteria shown in the table below based on the mechanistic modeling results.

In the Lower St. Johns River water clarity endpoints were available, so proposed criteria were derived that were protective of chl-a associated with balanced phytoplankton biomass and DO concentrations sufficient to maintain aquatic life. For this system, EPA used the DO from the SSAC, developed by FDEP and SJRWMD (2006) and adopted for the marine portion of the Lower St. Johns River, as an alternate DO endpoint with which to derive the proposed criteria to support DO concentrations sufficient to maintain aquatic life. This DO criterion, adopted as a water quality standard specific to this system, was used as an alternative target to the daily water column average DO concentration of 5.0 mg/L.

Through evaluation of chl-a and DO targets, EPA found that both targets were met under the 1995–1999 loads (see Volume 1, Appendix G). The values under mechanistic modeling represent the 90th percentile annual geometric mean nutrient concentrations from the 1995–1999 modeled nutrient loads.

Using statistical models, EPA developed candidate criteria for chl-a in all segments of the Lower St. Johns River. EPA also developed candidate criteria for TN and TP in segment 1803 using statistical models. Criteria derived from statistical analysis corroborate the criteria derived from the mechanistic model.

Proposed numeric nutrient criteria for TN, TP, and chl-a in St. Johns River segments are summarized in Table 2-161.

Table 2-161. Proposed and candidate numeric nutrient criteria for St. Johns River segments

		Proposed Criteria			Mechanistic Modeling			Statistical Modeling		
SEGMENT	SEGMENT ID	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
Lower St. Johns River	1801	0.75	0.095	2.5	0.75	0.095	2.5	-	-	6.1
Middle St. Johns River	1802	1.09	0.108	3.6	1.09	0.108	3.6	-	-	8.5
Upper St. Johns River	1803	1.15	0.074	7.7	1.15	0.074	7.7	0.96	0.11	8.4

2.21.8. Downstream Protective Values

In the St. Johns River mechanistic models were applied to derive proposed DPVs for TN and TP shown in Table 2-162.

Table 2-162. Proposed DPVs for St. Johns River

Tributary	LSPC Model Watershed ID	USGS Reach Code		TN (mg/L)	TP (mg/L)
Dunn Creek	N/A	03080103000060	1802	0.96	0.257
Broward River	N/A	03080103002013	1802	1.41	0.430
Unnamed Creek	N/A	03080103002589	1801	0.82	0.154
Mt Pleasant Creek	N/A	03080103002657	1801	0.85	0.235
Drummond Creek	N/A	03080103002002	1802	1.79	0.492
Unnamed Creek	N/A	03080103001496	1802	0.97	0.383
New Castle Creek	N/A	03080103001495	1802	0.92	0.334
Jones Creek	N/A	03080103001492	1801	0.95	0.394
Cow Head Creek	N/A	03080103002664	1801	0.94	0.371
Gin House Creek	N/A	03080103001488	1801	0.94	0.380
Pablo Creek (ICW)	N/A	03080103002626	1701	1.03	0.314
Trout River	N/A	03080103000066	1802	1.56	0.427
West Branch	N/A	03080103001997	1802	0.91	0.465
Block House Creek	N/A	03080103001999	1802	1.22	0.455
Ribault River	N/A	03080103000076	1802	1.25	0.399
Moncrief Creek	N/A	03080103001967	1802	1.03	0.326
Long Branch	N/A	03080103001966	1802	1.01	0.479
Pottsburg Creek	N/A	03080103000272	1802	1.03	0.436
Miller Creek	N/A	03080103001505	1802	1.03	0.545
Hogan Creek	N/A	03080103001146	1802	1.04	0.557
McCoy Creek	N/A	03080103001148	1802	1.02	0.534
Cedar River	N/A	03080103002468	1803	0.96	0.369
Williamson Creek	N/A	03080103002593	1803	1.04	0.569
Craig Creek	N/A	03080103002668	1803	1.04	0.533
Unnamed Creek	N/A	03080103002669	1803	1.04	0.566
Butcher Pen Creek	N/A	03080103001953	1803	1.04	0.531
Big Fishweir Creek	N/A	03080103002706	1803	1.03	0.547
Fishing Creek	N/A	03080103002703	1803	0.98	0.401
New Rose Creek	N/A	03080103001511	1803	1.02	0.523
Ortega River	N/A	03080103000085	1803	1.12	0.336
Goodbys Creek	N/A	03080103002671	1803	0.99	0.419
St Johns River	N/A	03080103000030	1803	1.12	0.070

2.21.9. Alternate Analysis

2.21.9.1. Lower St. Johns River

Criteria are in effect for the Lower St. Johns River to support TMDL and SSAC activities in the river. The TMDL was developed by FDEP in cooperation with the SJRWMD as part of its development of Pollutant Load Reduction Goals (PLRGs). Because of the river's impairment, SJRWMD and the U.S. Army Corps of Engineers began to develop a watershed model to estimate nonpoint source loads and to develop a linked hydrologic/water quality model to determine the assimilative capacity of the river. As a Class III water body, water quality criteria applicable to the impairment addressed by the TMDL are both the Florida DO criterion and the narrative nutrient criterion. The SSAC, a minimum DO concentration of 4 mg/L with a minimum daily average of 5 mg/L, was developed in the marine portion of the river. In addition, the Total

Fractional Exposure to DO levels in the 4.0 to 5.0 mg/L range must also be at or below 1.0 mg/L for each annual evaluation period as described by the equation in section 2.21.5.3. EPA approved it in 2006, and it is in effect as a water quality standard. The TMDL established numeric criteria for TN and TP loads in the freshwater portion of the Lower St. Johns River and TN load criteria in the saline portion to achieve the marine DO SSAC and protect the freshwater section for DO. The year 1999 was selected as the period to establish nitrogen load reductions to protect ecological health from large fish kills and low DO levels that occurred in the year (Henderickson et al. 2003).

Similar to the modeling approach proposed by EPA for Florida estuaries, TN, TP, and chl-a criteria were derived for the Lower St. Johns River using linked hydrologic (Pollutant Load Simulation Model [PLSM]), hydrodynamic (Environmental Fluid Dynamics Code [EFDC]), and water quality (three-dimensional eutrophication model [CE-QUAL-ICM]) models. Nutrient loading from the watershed to the Lower St. Johns River was estimated for point sources using National Pollutant Discharge Elimination System (NPDES) permits. Nonpoint nutrient inputs from the watershed to the river were estimated for each subbasin in the Lower St. Johns River using the PLSM (Adamus and Bergman 1995; Hendrickson and Konwinski 1998), estimates of atmospheric deposition, and estimates of loading from tributaries and upstream. PLSM was used to estimate seasonal loading from the watershed according to land use and runoff volume. Details of PLSM setup and calibration are in Hendrickson and Konwinski (1998) and Hendrickson et al. (2002). Within the river, hydrodynamics were modeled using an EFDC model and water quality processes were modeled using the U.S. Army Corps of Engineers Quality Integrated Compartment Model (CE-QUAL-ICM), Version 2 (Cerco and Cole 1993). The models were calibrated for the period from January 1, 1995, to November 30, 1998. The details of model setup and calibration for EFDC are in Sucsy and Morris (2002), and details of CE-QUAL-ICM are in Sucsy and Hendrickson (2003) and Tillman et al. (2004). Documentation of SJRWMD-proposed numeric nutrient criteria and methods for their derivation can be found in Volume 1, Appendix G.

2.21.10. References

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2.22. Nassau River/Big Talbot

2.22.1. Proposed Numeric Nutrient Criteria

Proposed numeric nutrient criteria for TN, TP, and chl-a in Nassau River/Big Talbot segments are summarized in Table 2-163.

Table 2-163. Proposed numeric nutrient criteria for Nassau River/Big Talbot segments

Segment Name	Segment Number	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
Lower Nassau	1901	0.33	0.113	3.2
Middle Nassau	1902	0.40	0.120	2.4
Upper Nassau	1903	0.75	0.125	3.4

2.22.2. General Characteristics

2.22.2.1. System Description

The Nassau River watershed, in Nassau County and part of Duval County, drains about 464 mi^2 (1,202 km^2) with 55 river miles (89 km) and 10 mi^2 (26 km^2) of estuary. The estuary is described as a series of sea islands that stretch along the Atlantic coastline (FDEP 2007).[101] The vast salt marsh estuary encompasses numerous interconnecting tidal creeks, channels, and tree islands, which support recreational and commercial fishing (FDEP 2007, 2011a).

Much of the lower portion of the watershed is part of the 108-mi^2 (279-km^2) Nassau River–St. Johns River Marshes Aquatic Preserve (FDEP 2007, 2011b). The lower portion of the watershed also composes the northern two-thirds of the federal Timucuan Ecological and Historic Preserve, which covers about 72 mi^2 (186 km^2) between the St. Johns and Nassau rivers (DeVivo et al. 2009). A number of areas in the Nassau River watershed are recognized as OFWs.[102] In addition, one of the largest remaining areas of contiguous coastal uplands in Duval County is protected by the Pumpkin Hill Creek State Buffer Preserve (FDEP 2011a).

The climate of the Nassau River watershed is considered temperate, although precipitation trends are more typical of a tropical climate (FDEP 2007, 2010). The average annual precipitation in the watershed is approximately 53 in (135 cm) (FDEP 2010), with a rainy season typically extending from June to October and the heaviest rainfall generally occurring in the late summer (FDEP 2007).

The topography of Nassau–St. Marys River watershed is low and flat and generally rises inland (FDEP 2007, 2010). Nassau Estuary and Talbot Island State Parks are at the southern end of a long chain of sea islands (FDEP 2007). Those islands and associated marshes, channels, and tributaries make up the southern extension of the St. Marys Meander Plain physiographic zone

[101] The information presented in this system description was compiled to summarize local information pertaining to the proposed numeric nutrient criteria for this estuary. For more information on EPA's process of delineating, segmenting, and deriving numeric nutrient criteria for this estuary, please see Section 1.3.
[102] Section 62-302.700 F.A.C.

(White 1970). The soils of the coastal lowland areas are typically sandy and poorly drained (FDEP 2010).

The watershed has minimal development, with small population centers in Yulee and Callahan and more concentrated urban development in the Amelia Island region (FDEP 2007). Forty-two percent of the watershed is forested, and 34 percent of the watershed is wetlands. Approximately 10 percent of the watershed is developed and predominantly clustered in small groupings throughout the watershed (Fry et al. 2011; SJRWMD 2006).[103] Silviculture is a common practice (FDEP 2007, 2010).

The Nassau River watershed is divided into two hydrological sections, an estuarine area and a riverine portion (FDEP 2007). In the estuarine area, there is a vast salt marsh estuary with many interconnecting tidal creeks and channels, as well as minor uplands (tree islands). The upper, riverine portion of the watershed has not been extensively documented (FDEP 2007). Approximately 44.3 mi^2 (114.7 km^2) of the Nassau–St. Marys River watershed is composed of surface waters including lakes, streams, wetlands, and springs that collectively compose about 3 percent of the total area (FDEP 2007). The main drainage features of the watershed are Nassau River and its tributaries. The Nassau–St. Marys watershed overlies all three principal Florida aquifer systems—surficial, intermediate, and Floridan (FDEP 2007).

2.22.2.2. Impaired Waters[104]

Three Class III marine WBIDs in the Nassau River Estuary have been listed for a nutrient-related parameter on Florida's CWA section 303(d) list approved by EPA. Of the three Class III marine WBIDs, one is impaired for DO (WBID 2129), and two are impaired for nutrients, chl-a, and DO (WBIDs 2130 and 2148B). No Class II WBIDs with nutrient-related impairments are documented for this area.[105]

No Class II or Class III marine WBIDs with nutrient-related TMDLs are documented in this region.[106]

[103] See Volume 1, Appendix C and its attachments for a summary of how land use data was combined and for a detailed breakdown of land uses.
[104] For a more detailed summary of this water body, see Volume 1, Appendix A.
[105] The nutrient-related 303(d) list was a compilation of EPA-approved 303(d) listing information for nutrients, chl-a, and DO provided in three decision documents: September 2, 2009, Basin Groups 1, 2, and 5 EPA Decision Document (http://www.epa.gov/region4/water/tmdl/florida/documents/fl09303d_decisiondoc_090209.pdf);
May 13, 2010, Basin Group 3 EPA Decision Document (http://www.epa.gov/region4/water/tmdl/florida/documents/fl303d_%20partialapproval_decision_docs051410.pdf);
and the December 21, 2010, Basin Group 4 EPA Decision Document (http://www.epa.gov/region4/water/tmdl/florida/documents/group_4_final_dec_doc_and_partial_app_letter_12_21_10.pdf).
[106] TMDLs were identified in February 2011 by compiling nutrient-related draft/final TMDLs from the following three sources:
FDEP TMDL website (http://www.dep.state.fl.us/water/tmdl/index.htm)
EPA Region 4 website (http://www.epa.gov/region4/water/tmdl/florida/index.html)
EPA National WATERS expert query tool (http://www.epa.gov/waters/tmdl/expert_query.html).

2.22.2.3. Water Quality

A recent evaluation of water quality in the Nassau River watershed is provided in the *Assessment of Coastal Water Quality at Timucuan Ecological and Historic Preserve, 2008*. Thirty sites in the preserve were randomly selected for monitoring, seventeen of which were in the Nassau River watershed. During the study in July 2008, DO concentrations from five stations located in Nassau Sound ranged from 5.33 to 7.01 mg/L (DeVivo et al. 2009). In another study, data collected between March 2004 to March 2005 from a sampling site at the confluence of the Ft. George River and Mud River in the Timucuan Ecological and Historic Preserve showed the minimum percent DO saturation (5.4%) in July and the highest (139.3%) in January. During the study period, several hypoxic events (DO saturation less than 28%) were observed. Those events occurred primarily in the summer and fall and were short-lived (none longer than 12 hours) and rare (only observed in 6% of deployments) (DiDonato et al. 2005).

Chl-a concentrations taken from five monitoring stations in Nassau Sound from DeVivo et al.'s 2008 study ranged from 2.12 to 9.75 µg/L, including two sampling sites with chl-a levels below 5 µg/L (2.12 and 2.48 µg/L) (DeVivo et al. 2009).

Livingston et al. (2002) reported periodic reductions in light transmission (associated with increases in water color) in the upper areas of the Nassau River Estuary during two year-long sampling periods of 1994 to 1995 and 1997 to 1998.

TDN concentrations, observed during a July 2008 assessment, were lowest at the two stations closest to the Atlantic Ocean, with concentrations of 0.092 and 0.081 mg/L (DeVivo et al. 2009).

TDP concentrations, observed during the July 2008 assessment from five monitoring sites in Nassau Sound, ranged from 0.023 to 0.032 mg/L (DeVivo et al. 2009). As of 2007, the median groundwater phosphorus concentration was 0.12 mg/L in the portion of the surficial aquifer that is in the Nassau River Planning Unit. Concentrations in the aquifer in this planning unit are high compared to statewide medians and with the other planning units in the watershed. It is unknown whether those concentrations are natural, anthropogenic, or both (FDEP 2007).

2.22.2.4. Biological Characteristics

The Nassau River watershed is largely undeveloped, with upland forests and wetlands covering the majority of the land (FDEP 2011a). The Nassau River Estuary, particularly the Nassau River–St. Johns River Marshes Aquatic Preserve and the Timucuan Ecological and Historic Preserve, are dominated by salt marsh (FDEP 2011b; FDNR 1986). Oyster bars, tidal flats, and tidal beaches are also important components of the watershed (FDEP 2010; FDNR 1986). Those dynamic communities form refugia, nursery grounds, and feeding areas for many other estuarine organisms, such as crabs and amphipods (FDEP 2010). The Nassau Wildlife Management Area covers 63 mi^2 (163 km^2 [40,168 ac]) along the upper river and tributaries of the watershed (FDEP 2011a).

Seagrass growth is reportedly not supported in this area because of a lack of suitable habitat (Sargent et al. 1995). However, salt marshes (especially tidal flats) in and around the Nassau River Estuary and watershed support a rich algal flora dominated by sea lettuce (*Ulva lactuca*). Lying on the surface layers of the sediments, algal growth is rapid and occurs throughout the year, contributing to the total primary production of the salt marsh ecosystem (FDNR 1986). No information was found regarding the presence of macroalgal blooms in Nassau Sound.

Phytoplankton abundance in the Nassau River Estuary is seasonally variable, with highest concentrations typically occurring during the summer and correlating with decreases in zooplankton. Large winter blooms of *Skeletonema costatum* appeared during 2000–2001 (Livingston et al. 2002).

The extensive estuarine area provides important habitat for a wide variety of invertebrates (e.g., penaeid shrimp, blue crabs, amphipods) throughout the Nassau River Estuary and watershed (FDNR 1986). Extensive shellfish beds are present around the mouth of the river (Boning 2007).

For a more detailed summary of this water body, see Volume 1, Appendix A.

2.22.3. Data Used

Several data sources specific to the Nassau River/Big Talbot were used in addition to those sources described in Section 1.4.3, as summarized in Table 2-164.

Table 2-164. Data sources specific to Nassau River/Big Talbot models

Data	Source	Location Used
Hydrologic group soils data	St. Johns River Water Management District (SJRWMD 2011)	St. Marys and Nassau watershed models
FDEP Level III Florida Land Use	St. Johns River Water Management District (SJRWMD 2006)	St. Marys and Nassau watershed models
Salinity, Temperature, and Water Quality Data	City of Jacksonville (FDEP 2007)	St. Marys-Nassau Estuary model
Municipal and industrial point sources	Georgia Environmental Protection Division (GAEPD 2008)	St. Marys-Nassau Estuary model
Climate data	Florida State Climate Center (FSCC 2009) and EarthInfo 2009 (EarthInfo 2009)	All watershed models
Municipal and industrial point sources	Georgia Environmental Protection Division (GAEPD 2008)	St. Marys watershed model
Water quality data	Georgia Environmental Protection Division (GAEPD 2008)	St. Marys watershed model
Georgia Land Use Trends land use	Natural Resources Spatial Analysis Lab (NARSAL 2008)	St. Marys watershed model

2.22.4. Segmentation

The GIS isohaline analysis and geomorphological structure yielded three segments for the Nassau Estuary. Figure 2-65 shows the resulting three segments for Nassau River/Big Talbot.

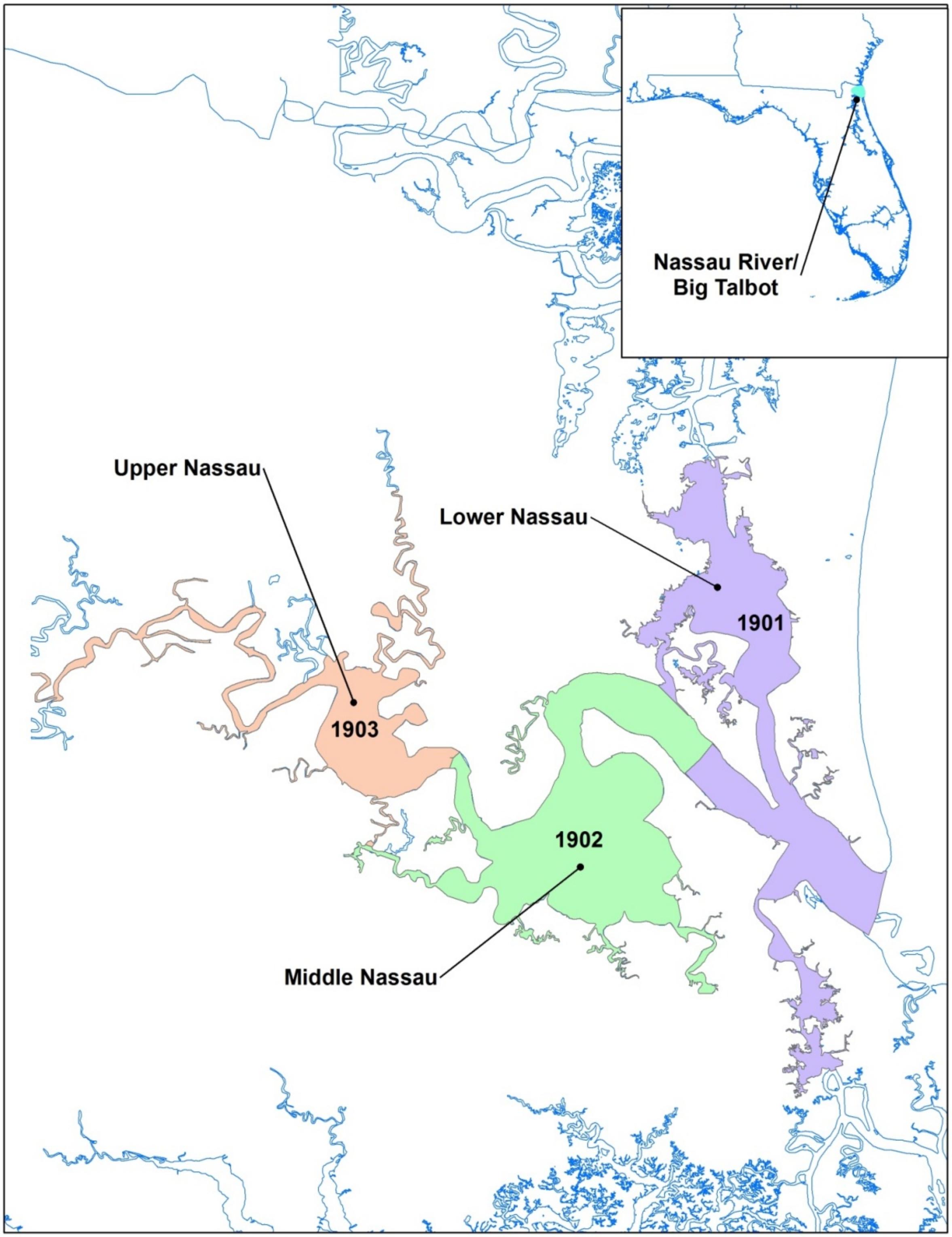

Figure 2-65. Results of Nassau River/Big Talbot Estuary segmentation

2.22.5. Water Quality Targets

2.22.5.1. Seagrass Depth and Water Clarity Targets

Seagrass is not historically present in the Nassau River/Big Talbot Estuary.

2.22.5.2. Chlorophyll a Target

To prevent nuisance algal blooms and protect the estuary's designated uses, chl-a levels must not exceed 20 µg/L more than 10 percent of the time. The rationale for that target is provided in Section 1.2.2.

2.22.5.3. Dissolved Oxygen Targets

On the basis of the rationale that sufficient DO is necessary to protect aquatic life, as described in Section 1.2.3, the following DO targets were established:

- Minimum allowable DO of 4.0 mg/L as a water column average for each estuary segment 90 percent of the time over the simulation's time span
- Daily average DO of 5.0 mg/L as a water column average for each estuary segment 90 percent of the time over the simulation's time span
- Minimum 3-hour average DO of 1.5 mg/L in the bottom two layers for each estuary segment over the simulation's time span

2.22.6. Results of Analyses

2.22.6.1. Mechanistic Model Analysis

Average load contributions from the Nassau watershed are shown in Table 2-165.

Table 2-165. Average load contributions from the Nassau watershed (2002–2009)

	TN Load[1] (kg/d; Mean ± se)			TP Load[1] (kg/d; Mean ± se)		
Year	Existing[2]	Background[3]	Anthropogenic[4]	Existing[2]	Background[3]	Anthropogenic[4]
2002	931 ± 116	673 ± 93	258 ± 24	83 ± 15	67 ± 14	16 ± 1
2003	760 ± 86	534 ± 63	226 ± 23	54 ± 7	40 ± 6	13 ± 1
2004	1,669 ± 192	1,275 ± 156	393 ± 37	143 ± 21	123 ± 20	20 ± 1
2005	1,511 ± 213	1,127 ± 177	384 ± 38	114 ± 25	94 ± 23	20 ± 2
2006	392 ± 44	273 ± 32	119 ± 12	30 ± 4	22 ± 4	9 ± 1
2007	555 ± 64	382 ± 47	173 ± 17	51 ± 8	39 ± 7	11 ± 1
2008	1,272 ± 202	950 ± 164	322 ± 39	101 ± 21	85 ± 20	16 ± 2
2009	1,267 ± 197	940 ± 154	328 ± 43	90 ± 19	74 ± 18	16 ± 2

[1] Annual average daily load from all terminal reaches
[2] Annual average daily load from all terminal reaches computed from existing model scenario
[3] Annual average daily load from all terminal reaches computed from background model scenario (remove anthropogenic sources/land use; use existing hydrology)
[4] Anthropogenic load computed as difference between existing and background load

The chl-a target was met for all Nassau River/Big Talbot segments on the basis of 2002–2009 nutrient loads. DO targets were not met for any segments. Those segments could not meet the DO targets using either the 2002–2009 nutrient loads scenario or the non-anthropogenic nutrient scenario. Table 2-166 identifies which targets were met under 2002–2009 nutrient loads and which targets were not met.

Sensitivity analyses revealed that DO was insensitive to changes in nutrients.

Table 2-166. Water quality targets met for Nassau River/Big Talbot based on mechanistic modeling

Segment	DO	Chl-a	K_d
1901	No	Yes	No target
1902	No	Yes	No target
1903	No	Yes	No target

A summary of candidate criteria for Nassau River/Big Talbot segments is given in Table 2-167. Nutrient loads from 2002–2009 were used to calculate candidate criteria for Nassau River/Big Talbot.

Table 2-167. Summary of candidate criteria for Nassau River/Big Talbot derived from mechanistic modeling

Segment	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
1901	0.33	0.113	3.2
1902	0.40	0.120	2.4
1903	0.75	0.125	3.4

2.22.6.2. Statistical Model Analysis

Data were not sufficient within each segment to conduct statistical analyses in Nassau River/Big Talbot.

2.22.7. Application of Analyses for Proposed Numeric Nutrient Criteria

EPA evaluated various lines of evidence for deriving criteria in Nassau River/Big Talbot Estuary. Data necessary to conduct statistical analyses were not sufficient. EPA derived the proposed numeric nutrient criteria shown in the table below based on the mechanistic modeling results.

In this estuary seagrass has not been historically present, so EPA evaluated two endpoints in the mechanistic modeling approach for Nassau River: (1) chl-a concentrations associated with balanced phytoplankton biomass, and (2) DO concentrations sufficient to maintain aquatic life. EPA found that the DO target was not met under the 2002–2009 loads and was insensitive to changes in nutrients. Therefore, the DO target was not used in the Nassau River system. The proposed criteria were derived to be protective of chl-a concentrations. The values under mechanistic modeling represent the 90th percentile annual geometric mean nutrient concentrations from the 2002–2009 modeled nutrient loads.

Since the mechanistic model provided values for every segment in the estuary, EPA derived the proposed numeric nutrient criteria for Nassau River/Big Talbot shown in Table 2-168.

Table 2-168. Proposed and candidate numeric nutrient criteria for Nassau River/Big Talbot segments

		Proposed Criteria			Mechanistic Modeling			Statistical Modeling		
SEGMENT	SEGMENT ID	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
Lower Nassau	1901	0.33	0.113	3.2	0.33	0.113	3.2	-	-	-
Middle Nassau	1902	0.40	0.120	2.4	0.40	0.120	2.4	-	-	-
Upper Nassau	1903	0.75	0.125	3.4	0.75	0.125	3.4	-	-	-

2.22.8. Downstream Protective Values

In Nassau River/Big Talbot mechanistic models were applied to derive the proposed DPVs for TN and TP shown in Table 2-169.

Table 2-169. Proposed DPVs for Nassau River/Big Talbot

Tributary[a]	LSPC Model Watershed ID	USGS Reach Code	Estuary Segment	TN (mg/L)	TP (mg/L)
Lofton Creek	90001	03070205000014	1903	0.74	0.019
Pumpkin Hill Creek	90002	03070205000336	1902	0.49	0.015
Nassau River	90004	03070205000016	1903	1.06	0.018
	90008	03070205002555	1903	0.73	0.027
Thomas Creek	90010	03070205000037	1903	0.45	0.022
Nassau River	90012	03070205000022	1903	0.85	0.012
Mink Creek	90014	03070205000168	1902	1.17	0.024
Intracoastal Waterway	90018	03070205000330	1901	0.51	0.020
Nassau River	90022	03070205000025	1903	0.81	0.106

[a] Tributary names left blank are unnamed

2.22.9. References

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2.23. St. Marys River/Amelia River

2.23.1. Proposed Numeric Nutrient Criteria

Proposed numeric nutrient criteria for TN, TP, and chl-a in St. Marys River/Amelia River segments are summarized in Table 2-170.

Table 2-170. Proposed numeric nutrient criteria for St. Marys River/Amelia River segments

Segment Name	Segment Number	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
Lower St. Marys	2002	0.27	0.045	3.0
Middle St. Marys	2003	0.44	0.036	2.7

2.23.2. General Characteristics

2.23.2.1. System Description

The St. Marys watershed occupies 1,585 mi^2 (4,105 km^2) in northeastern Florida and southeastern Georgia. About 942 mi^2 (2,440 km^2) (59%) of the watershed is in Florida (FDEP 2011; SJRWMD 1993). The St. Marys River is a blackwater stream with an extensive freshwater and saltwater marsh system that flows along a twisting 130-mi (210-km) path into the Cumberland Sound and the Atlantic Ocean (Boning 2007; FDEP 2010a; SJRWMD 2011a). The river forms much of the border between northeast Florida and southeast Georgia (FDEP 2010b, 2011).[107] The river originates in the peat bogs of the Okefenokee Swamp in Georgia and discharges to the Cumberland Sound between Cumberland Island (Georgia) to the north and Amelia Island (Florida) to the south (FDEP 2007, 2011).

St. Marys River is used primarily for recreational and sightseeing purposes, because there are few river crossings and relatively little development along its banks (SJRWMD 2010). St. Marys Inlet is at the entrance to Cumberland Sound and the mouths of the St. Marys and Amelia rivers,

[107] The information presented in this system description was compiled to summarize local information pertaining to the proposed numeric nutrient criteria for this estuary. For more information on EPA's process of delineating, segmenting, and deriving numeric nutrient criteria for this estuary, please see Section 1.3.

and it serves as the major inlet for St. Marys Estuary (FDEP 2011). The inlet also provides an important passageway for commercial, recreational, and U.S. Navy vessels (FDEP 2007, 2010a, 2011). Several surface water bodies in the St. Marys watershed have been designated as OFWs, including Amelia Island State Recreational Area, Fort Clinch State Park, Fort Clinch Aquatic Preserve, and waters in the Osceola National Forest.[108]

The Nassau–St. Marys watershed is in the temperate/subtropical climate zone, but typically the climate is more tropical than temperate, with a rainy season (June through October) and dry season (November through May) (FDEP 2007, 2010a). Annual average precipitation in the watershed is approximately 53 in (135 cm), with the greatest rainfall generally occurring in late summer (FDEP 2010a). Floodplains cover approximately half of the lower St. Marys watershed (FDEP 2011). The low topography and slope of the lower watershed, combined with tidal effects, make lower St. Marys watershed a poorly drained area of the watershed (SJRWMD 1993). Tides are significant in this part of northeast Florida and can raise or lower water levels by as much as 5 ft (1.5 m) (Boning 2007).

The dominant land use in the St. Marys watershed is silviculture, and the primary land cover is upland forest, the latter being primarily managed pine forests (FDEP 2010b; SJRWMD 1993; SMRMC 2003). FDEP estimated that approximately 70 percent of the entire St. Marys watershed is in large-tract private ownership for silvicultural and conservation purposes (FDEP 2007). Future land use projections indicate the conversion of rangeland and forest lands into residential, commercial, and industrial land uses in the coastal regions of Nassau County (SJRWMD 2011a). Livestock and poultry operations, as well as commodity production, are the primary agricultural practices in the watershed (FDEP 2007, 2010a).

The surficial, intermediate, and Floridan aquifer systems are the three principal groundwater resources beneath the St. Marys watershed (FDEP 2007). The surficial aquifer system is the water table aquifer and is used as a potable water supply to a limited extent. It is important because it directly interacts with surface water bodies, providing baseflow to streams, estuaries, and lakes (FDEP 2010b). The intermediate aquifer system is composed of clays and acts mainly as a confining layer. The Floridan aquifer is deep, confined, and under artesian pressure and serves as the primary source of potable water in the watershed (FDEP 2007, 2010b).

2.23.2.2. Impaired Waters[109]

One Class III marine water body identification number (WBID) in St. Marys Estuary has been listed for a nutrient-related parameter on Florida's CWA section 303(d) list approved by EPA. The Class III marine WBID is impaired for DO (WBID 2097C). No Class II WBIDs with nutrient-related impairments are documented for the area.[110] No Class II or Class III marine WBIDs with nutrient-related TMDLs are documented in the region.[111]

[108] Section 62-302.700 F.A.C.
[109] For more information about the data source, see Volume 1, Appendix A.
[110] The nutrient-related 303(d) list was a compilation of EPA-approved 303(d) listing information for nutrients, chl-a, and DO provided in three decision documents: September 2, 2009, Basin Groups 1, 2, and 5 EPA Decision Document (http://www.epa.gov/region4/water/tmdl/florida/documents/fl09303d_decisiondoc_090209.pdf);
May 13, 2010, Basin Group 3 EPA Decision Document

2.23.2.3. Water Quality

Point and nonpoint sources of pollution in the combined Nassau–St. Marys watershed include stormwater runoff, silviculture, urban/suburban land uses, agricultural pollutants, and leaking septic tanks (FDEP 2010a; SJRWMD 2011a). The primary concern for the water quality of the St. Marys and other coastal rivers is secondary impacts from development such as leaking septic tanks and chemical and pesticide runoff from lawns and streets (SJRWMD 2011a).

Livingston et al. (2002) summarized a series of studies conducted in the Amelia River Estuary and compared results to reference measurements from the Nassau River Estuary. The study evaluated the effects of ammonia in pulp mill effluent discharging to the Amelia River on estuarine plankton and other biological communities (e.g., fish, invertebrates) from 1994 to 1995 and 1997 to 1998. A related laboratory mesocosm and microcosm study was also conducted to assess parameters related to primary production (e.g., chl-a, nutrients, turbidity, salinity) (Livingston et al. 2002).

St. Marys River has naturally high color and occasionally has relatively low DO in the summer because of accelerated decomposition in adjacent swamps (SJRWMD 1993; SMRMC 2003). FDEP's 2007 Status Monitoring Network assessment indicated that DO was below the Florida standard[112] in 20 percent of the Nassau–St. Marys watershed's rivers (including the north prong of St. Marys Estuary and St. Marys River) and 87 percent of the sampled streams (i.e., the remaining streams and tributaries) (FDEP 2010b). On the basis of FDEP IWR database Run 40 data, the Amelia River had somewhat lower DO than the lower St. Marys River portion of the estuary. Annual geometric means for DO ranged from 4.71 to 8.77 mg/L.

Chl-a data from FDEP's IWR Run 40 database of the St. Marys River showed geometric mean concentrations ranging from 0.61 to 2.67 μg/L between 1987 and 2004. Overall, there is little data for chl-a and Bricker et al. (2007) reported that, as of 2004, the data were too limited to provide an overall assessment of chl-a in St. Marys River and Cumberland Sound.

Turbidity data from FDEP's IWR Run 40 database for two locations in the St. Marys/Amelia River Estuary were variable, and the geometric means ranged from 3.0 to 33.0 NTU between 1979 and 2004.

Annual geometric means of TN and TP concentrations from FDEP's IWR Run 40 database for estuarine segments of the Lower St. Marys River ranged from 0.19 to 0.78 mg/L and 0.028 to 0.230 mg/L, respectively, between 1979 and 2004. TP concentrations appeared to be higher in the Amelia River compared to the St. Marys River segments of the estuary.

(http://www.epa.gov/region4/water/tmdl/florida/documents/fl303d_%20partialapproval_decision_docs051410.pdf); and the December 21, 2010, Basin Group 4 EPA Decision Document (http://www.epa.gov/region4/water/tmdl/florida/documents/group_4_final_dec_doc_and_partial_app_letter_12_21_10.pdf).

[111] TMDLs were identified in February 2011 by compiling nutrient-related draft/final TMDLs from the following three sources:
FDEP TMDL website (http://www.dep.state.fl.us/water/tmdl/index.htm)
EPA Region 4 website (http://www.epa.gov/region4/water/tmdl/florida/index.html)
EPA National WATERS expert query tool (http://www.epa.gov/waters/tmdl/expert_query.html).

[112] The Florida DO standard states that DO "shall not average less than 5.0 (mg/L) in a 24-hour period and shall never be less than 4.0 (mg/L). Normal daily and seasonal fluctuations above these levels shall be maintained" (Section 62-302.530, F.A.C.).

2.23.2.4. Biological Characteristics

The St. Marys watershed has an extensive variety of types of wetlands that are fundamental to maintaining flows and water quality in the river and are an important component in the diverse habitats in the watershed (SJRWMD 1993).

No active seagrass monitoring is underway for northeast Florida estuaries (including the St. Marys and Amelia estuaries) (FFWCC 2011). Livingston et al. (2002) described the impact of ammonia from pulp mill discharges on phytoplankton assemblages between 1994–1995 and 1997–1998. By comparing the Amelia River (affected by pulp mill effluent) with the nearby Nassau River (with no pulp mill discharge) and using mesocosm studies, Livingston et al. (2002) demonstrated that ammonia reduced the abundance and species diversity of phytoplankton. Livingston et al. (2002) found that phytoplankton numbers in the Amelia system were only about 57 percent of those found in the Nassau system. A documented HAB caused by *K. brevis* originated near the mouth of the Amelia River, affecting Fernandina Beach in September of 2007, and was later observed at sites up to 200 mi (320 km) south of the Fernandina Beach site (Reich et al. 2008).

A wide variety of habitats support many invertebrates in the St. Marys watershed (FDNR 1986). More than 65 species of fish have been identified in the St. Marys River (Boning 2007; SJRWMD 2011a). Fort Clinch State Park Aquatic Preserve and the larger Nassau River–St. Johns River Marsh Aquatic Preserve to the south provide protected fisheries habitat for spawning and juvenile development (FDNR 1986).

For a more detailed summary of this water body, see Volume 1, Appendix A.

2.23.3. Data Used

Several data sources specific to the St. Marys River/Amelia River were used in addition to those sources described in Section 1.4.3, as summarized in Table 2-171.

Table 2-171. Data sources specific to St. Marys River/Amelia River models

Data	Source	Location Used
Hydrologic group soils data	St. Johns River Water Management District (SJRWMD 2011b)	St. Marys and Nassau watershed models
FDEP Level III Florida Land Use	St. Johns River Water Management District (SJRWMD 2006)	St. Marys and Nassau watershed models
Salinity, Temperature, and Water Quality Data	City of Jacksonville (FDEP 2007)	St. Marys-Nassau Estuary model
Municipal and industrial point sources	Georgia Environmental Protection Division (GAEPD 2008)	St. Marys-Nassau Estuary model
Climate data	Florida State Climate Center (FSCC 2009) and EarthInfo (EarthInfo 2009)	All watershed models
Municipal and industrial point sources	Georgia Environmental Protection Division (GAEPD 2008)	St. Marys watershed model
Water quality data	Georgia Environmental Protection Division (GAEPD 2008)	St. Marys watershed model
Georgia Land Use Trends land use	Natural Resources Spatial Analysis Lab (NARSAL 2008)	St. Marys watershed model

2.23.4. Segmentation

The isohaline GIS analysis of St. Marys Estuary yielded four distinct segments: Cumberland Sound, Upper, Middle, and Lower St. Marys. Because Cumberland Sound and Upper St. Marys are in Georgia, they were removed from the segmentation scheme. The Middle St. Marys segment was also cut at the Georgia-Florida state line to exclude portions in Georgia. Figure 2-66 shows the resulting two segments for St. Marys River/Amelia River.

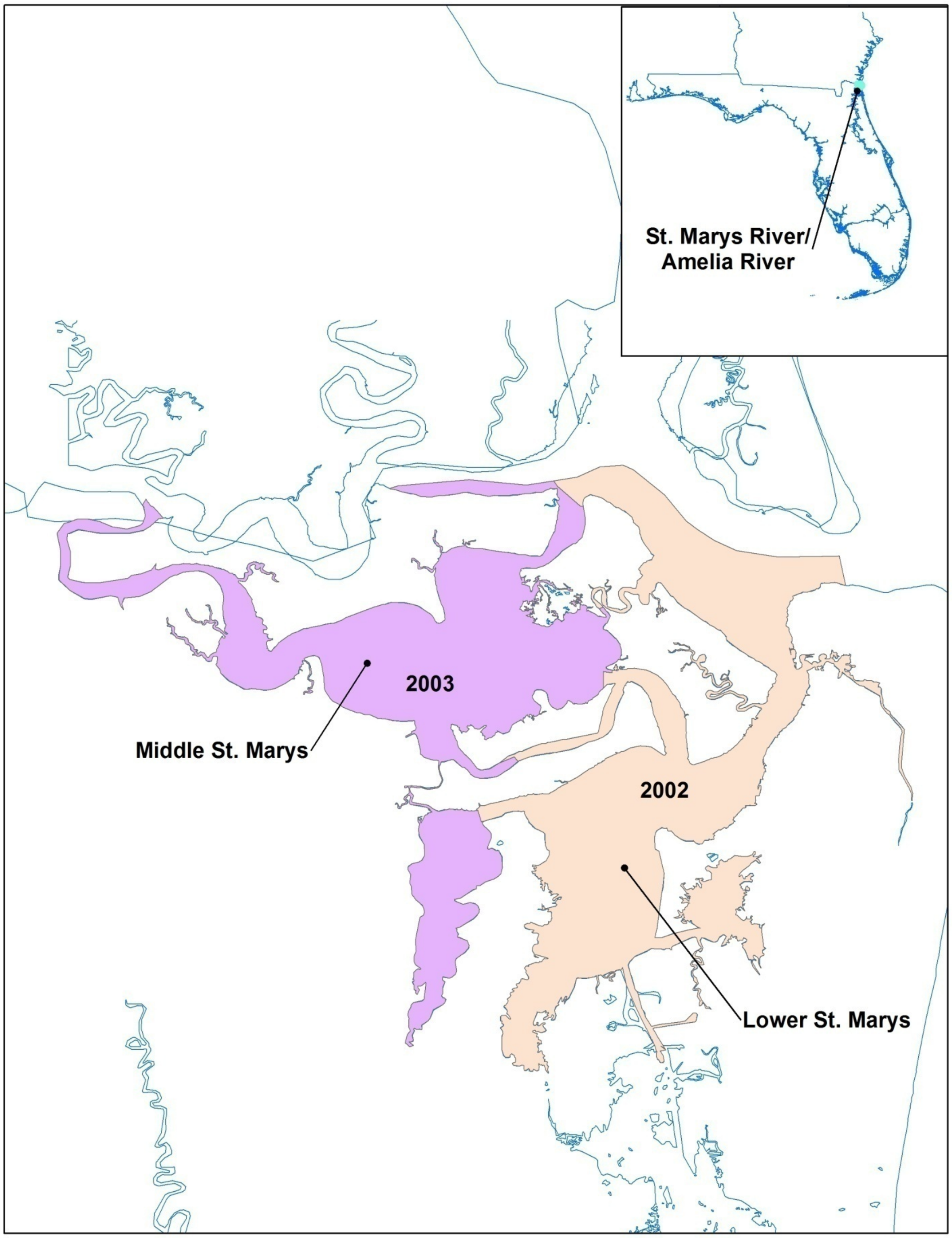

Figure 2-66. Results of St. Marys River/Amelia River segmentation

2.23.5. Water Quality Targets

2.23.5.1. Seagrass Depth and Water Clarity Targets

Seagrass is not historically present in the St. Marys River/Amelia River Estuary.

2.23.5.2. Chlorophyll a Target

To prevent nuisance algal blooms and protect the estuary's designated uses, chl-a levels must not exceed 20 µg/L more than 10 percent of the time. The rationale for that target is provided in Section 1.2.2.

2.23.5.3. Dissolved Oxygen Targets

On the basis of the rationale that sufficient DO is necessary to protect aquatic life, as described in Section 1.2.3, the following DO targets were established:

- Minimum allowable DO of 4.0 mg/L as a water column average for each estuary segment 90 percent of the time over the simulation's time span
- Daily average DO of 5.0 mg/L as a water column average for each estuary segment 90 percent of the time over the simulation's time span
- Minimum 3-hour average DO of 1.5 mg/L in the bottom two layers for each estuary segment over the simulation's time span

2.23.6. Results of Analyses

2.23.6.1. Mechanistic Model Analysis

Average load contributions from the St. Marys watershed are shown in Table 2-172.

Table 2-172. Average load contributions from the St. Marys watershed (2002–2009)

	TN Load[1] (kg/d; Mean ± se)			TP Load[1] (kg/d; Mean ± se)		
Year	**Existing[2]**	**Background[3]**	**Anthropogenic[4]**	**Existing[2]**	**Background[3]**	**Anthropogenic[4]**
2002	3,799 ± 266	3,162 ± 238	637 ± 30	93 ± 10	51 ± 9	43 ± 1
2003	7,808 ± 682	6,919 ± 628	890 ± 56	135 ± 10	79 ± 8	56 ± 2
2004	8,059 ± 866	7,128 ± 785	931 ± 84	167 ± 19	110 ± 16	57 ± 3
2005	9,952 ± 573	8,831 ± 517	1,121 ± 58	173 ± 14	114 ± 12	59 ± 2
2006	3,353 ± 367	2,936 ± 339	417 ± 30	71 ± 5	31 ± 4	40 ± 1
2007	2,963 ± 211	2,433 ± 181	529 ± 32	94 ± 7	44 ± 6	51 ± 2
2008	6,630 ± 815	5,874 ± 744	756 ± 75	140 ± 20	84 ± 17	56 ± 3
2009	5,859 ± 650	5,086 ± 582	773 ± 71	110 ± 11	61 ± 9	49 ± 2

[1] Annual average daily load from all terminal reaches
[2] Annual average daily load from all terminal reaches computed from existing model scenario
[3] Annual average daily load from all terminal reaches computed from background model scenario (remove anthropogenic sources/land use; use existing hydrology)
[4] Anthropogenic load computed as difference between existing and background load

The chl-a target was met for St. Marys River/Amelia River segments on the basis of 2002–2009 nutrient loads. DO targets were not met for any segments. Those segments could not meet the DO targets using either the 2002–2009 nutrient loads scenario or the non-anthropogenic nutrient scenario. Table 2-173 identifies which targets were met under 2002–2009 nutrient loads and which targets were not met.

Sensitivity analyses revealed that DO was insensitive to changes in nutrients.

Table 2-173. Water quality targets met for St. Marys River/Amelia River based on mechanistic modeling

Segment	DO	Chl-a	K_d
2002	No	Yes	No target
2003	No	Yes	No target

A summary of candidate criteria for St. Marys River/Amelia River segments is given in Table 2-174. Nutrient loads from 2002–2009 were used to calculate candidate criteria for St. Marys River/Amelia River.

Table 2-174. Summary of candidate criteria for St. Marys River/Amelia River derived from mechanistic modeling

Segment	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
2002	0.27	0.045	3.0
2003	0.44	0.036	2.7

2.23.6.2. Statistical Model Analysis

Data were not sufficient within each segment to conduct statistical analyses in St. Marys River/Amelia River.

2.23.7. Application of Analyses for Proposed Numeric Nutrient Criteria

EPA evaluated various lines of evidence for deriving numeric nutrient criteria in St. Marys River/Amelia River. There were insufficient data in the St. Marys River/Amelia River Estuary to derive the proposed criteria using statistical models. The proposed numeric nutrient criteria were derived using mechanistic modeling output.

In this estuary seagrass has not been present historically. As a result EPA, evaluated two endpoints for St. Marys River/Amelia River: (1) chl-a concentrations associated with balanced phytoplankton biomass, and (2) DO concentrations sufficient to maintain aquatic life. EPA found that the DO target was not met under the 2002–2009 loads and was insensitive to changes in nutrients. Therefore, the DO endpoint was not used in the St. Marys River/Amelia River system. The proposed criteria were derived to be protective of chl-a concentrations. The values under mechanistic modeling represent the 90th percentile annual geometric mean nutrient concentrations from the 2002–2009 modeled nutrient loads.

Proposed numeric nutrient criteria for TN, TP, and chl-a in St. Marys River/Amelia River segments are summarized in Table 2-175.

Table 2-175. Proposed and candidate numeric nutrient criteria for St. Marys River/Amelia River segments

		Proposed Criteria			Mechanistic Modeling		
SEGMENT	SEGMENT ID	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)	TN (mg/L)	TP (mg/L)	Chl-a (µg/L)
Lower St. Marys	2002	0.27	0.045	3.0	0.27	0.045	3.0
Middle St. Marys	2003	0.44	0.036	2.7	0.44	0.036	2.7

2.23.8. Downstream Protective Values

In St. Marys River/Amelia River mechanistic models were applied to derive the proposed DPVs for TN and TP shown in Table 2-176.

Table 2-176. Proposed DPVs for St. Marys River/Amelia River

Tributary	LSPC Model Watershed ID	USGS Reach Code	Estuary Segment	TN (mg/L)	TP (mg/L)
Amelia River	80011	03070204000709	2002	2.40	0.013
Saint Marys River	80019	03070204000143	2003	1.42	0.057

2.23.9. References

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3. Other Analyses: Tidal Creeks and Marine Lakes

3.1. Tidal Creeks

Tidal creeks are small subestuaries that exhibit a wide range of salinity zones typical of larger estuaries, but on a smaller scale. They are formed by typical estuarine processes, combining tidal and freshwater flows, and are intimately connected to benthic processes because of their small size and typically shallower depths. They can be fed by a freshwater stream, but smaller tidal creeks in extensive salt marsh and mangrove areas might have no surface inputs at all. Rather, they are simply conduits for tidal water and rainfall-produced runoff to enter and leave wetland areas. All tidal creeks receive some freshwater inputs via runoff and groundwater, and some exhibit substantial seepage during low tide.

Tidal action causes salinity changes at some locations in a creek such as at a bridge or other fixed monitoring station. Salinity is lower during ebb tides when freshwater flow from surface and groundwater dominate, and it is higher during rising tides when water from the main estuary dominates. The range of salinity fluctuation depends on location, amount of freshwater inflow, wind, and distance from a marine water source. Total suspended solids (TSS) and turbidity are generally higher in tidal creeks than in adjacent estuaries because they are shallow, often have exposed mud banks at low tide, and have high flow velocities that resuspend and transport fine sediment. Higher turbidity occurs in creeks with greater tidal ranges, such as St. Marys River on the Atlantic coast (2 m tidal range), compared to those with lower tidal ranges, such as St. Andrews Bay on the Gulf coast (0.5 m range) (Tides High and Low Inc. 2011).

Benthic and planktonic algae are the dominant primary producers in tidal creeks (Janicki Environmental 2011). Because tidal creeks are shallow, benthic primary production is more important than in open estuaries where phytoplanktonic production dominates (Janicki Environmental 2011). Water column chl-a concentrations are highly variable in tidal creeks depending on salinity and the type of phytoplankton species (freshwater or marine). Chl-a concentrations are often higher at low tide when freshwater is dominant (Mallin et al. 1999), but could be higher at high tide if the main bay has high concentrations of phytoplankton. Emergent vegetation (e.g., marsh grasses, mangroves) often thrive on the banks of tidal creeks because of the high nutrient flux and extensive root systems to hold them in place. Seagrasses generally do not occur in tidal creeks because they are sensitive to current scour and salinity fluctuations (e.g., Janicki Environmental 2011).

The water quality in tidal creeks is highly variable and often poorer than the main body of the estuary. Tidal creeks have higher concentrations of particulate and dissolved organic matter than adjacent open estuaries. In addition to benthic and planktonic algae, tidal creeks receive organic matter from salt marsh or mangrove vegetation along their banks and dissolved and particulate organic matter from upland freshwater portions of the creek. Thus, allochthonous organic matter plays a larger role in tidal creek metabolism than in adjacent larger estuaries (Janicki Environmental 2011). Because of the high organic matter loading and high temperatures in shallow water, episodes of hypoxia can occur even in undisturbed (natural) tidal creeks during summer months (Lerberg et al. 2000; MacPherson et al. 2007).

Tidal creeks and associated marshes and mangroves are refuges for small forage fish and for juveniles of larger fish to which they are considered an important spawning and nursery habitat. For example, juveniles of common snook (*Centropomus undecimalis*) depend on tidal creeks for shelter from larger predators (Adams 2005). Dominant aquatic animals in tidal creeks include mummichugs (Fundulidae) and grass shrimp (Palaeomonidae), but many other estuarine species also thrive in these habitats (Greenwood et al. 2009; Krebs et al. 2009; Janicki Environmental 2011). In general, undisturbed tidal creeks in Florida have higher fish densities than adjacent open waters.

Tidal creeks can be degraded by suburban and urban development in their watersheds. Stressors from watershed development include hydrologic modification because of increased flashiness from impervious surfaces; channelization for marinas and docks; and nutrient pollution from lawn fertilizers, urban and agricultural runoff, and septic systems. As a result, tidal creeks draining developed areas have higher nutrient, chlorophyll, and fecal coliform bacteria concentrations compared to streams draining undeveloped watersheds (Holland et al. 2004; Mallin et al. 2004). Furthermore, hypoxic episodes are more extreme (prolonged and with lower DO) in developed watersheds than in undeveloped watersheds (Holland et al. 2004).

In addition to increased nutrient concentrations, watershed development results in increased variability and volume of runoff during and after rainfall. The runoff surges cause more rapid and more extreme salinity changes as well as increased scour and changes in channel morphology. Tidal creeks with watersheds that have high impervious surface area have been observed to support degraded fish and invertebrate communities in South Carolina. Although commercially important spot and shrimp populations were reduced in affected creeks, mummichug and grass shrimp remained (Holland et al. 2004; Lerberg et al. 2000; Mallin et al. 2004). Other studies have shown that low-salinity waters of tidal creeks in developed areas can develop nuisance algal bloom conditions (Mallin et al. 2004; MacPherson et al. 2007), with the bloom waters moving back and forth with the tides. Such bloom conditions can also contribute to more severe hypoxic episodes.

3.1.1. Derivation of Numeric Nutrient Criteria for Tidal Creeks

Tidal creeks were classified separately from estuaries because tidal creeks are expected to have higher nutrient and chlorophyll concentrations than adjacent, open waters. The classification and segmentation approach used for estuaries was not considered practical because of the large number and variety of small systems. A definitional approach was chosen, applicable to all tidal creeks, to be implemented on a case-by-case basis as data allow.

Several options were considered for deriving numeric nutrient criteria for tidal creeks, including applying inland freshwater criteria derived for upstream waters or applying estuarine criteria derived for downstream waters. Neither of those two approaches alone would be applicable to the full range and variability of tidal creeks. Ultimately, EPA selected two approaches for deriving numeric TN and TP criteria that account for the inherent variability of tidal creeks.[1] The first approach is to apply separately derived inland TN and TP criteria for adjacent freshwaters if the mean chloride of the tidal creek is less than 1,500 mg/L, or apply estuarine TN and TP

[1] Neither approach supports derivation of chl-a criteria because of the variability of benthic algae in streams and uncertainty regarding expected levels of chl-a in tidal creeks.

criteria for adjacent downstream waters if the mean chloride of the tidal creek is greater than or equal to 1,500 mg/L.[2] The second approach uses linear interpolation to derive criteria for TN and TP for tidal creeks using criteria that were derived separately for adjacent inland freshwater and estuary areas on the basis of mean salinity. Criteria would be derived by that method only where there are sufficient salinity data to allow for interpolation. The calculation uses the following formula:

$$C_{TC} = C_{FW} + (S_{TC} - S_{FW}) \times \left(\frac{C_{Est} - C_{FW}}{S_{Est} - S_{FW}}\right)$$

where

C_{TC} = nutrient criterion for tidal creek segment

C_{FW} = nutrient criterion for adjoining/upstream freshwater segment

C_{Est} = nutrient criterion for adjoining estuarine segment

S_{TC} = mean salinity for tidal creek segment

S_{FW} = mean salinity for adjoining/upstream freshwater segment

S_{Est} = mean salinity for adjoining estuarine segment

Example:

Segment	Mean Salinity (ppt)	Criterion Concentration
Freshwater segment	0.5	2.5
Tidal creek segment	20	C_{TC}
Estuarine segment	30	0.8

$$C_{TC} = 2.5 + (20 - 0.5) \times \left(\frac{0.8 - 2.5}{30 - 0.5}\right) = 1.376$$

3.2. Marine Lakes

Marine lakes are coastal lakes with intermittent or groundwater connections to marine water. Many are small and shallow, and generally round or elliptical, reflecting their formation as depressions that became isolated from marine waters by sand and dune formation (FNAI 2010). They include dune lakes in Walton and Bay counties and solution lakes (also called rockland lakes) in Monroe County. Dune lakes are characterized by a sandy bottom, and solution lakes have a hardbottom formed by the dissolution of limestone. Solution lakes are often stratified by a salinity gradient, with a freshwater layer at the surface and a denser saline layer below. Some solution lakes are also meromictic (the layers rarely or never mix), and can have a naturally anoxic hypolimnion with characteristic chemosynthetic bacteria.

[2] The 1,500 mg/L chloride threshold is used to define waters as *predominantly freshwater* or *predominantly marine water* [F.A.C. 62-302.200(22) and 62-302.200(23)].

Lakes of marine origin are relatively closed systems especially when compared to the nearby marine or estuarine waters (Hutchinson 1957). Accordingly, the retention time of marine lakes is longer than estuaries and more similar to inland lakes. Retention time is a key factor in response to nutrient pollution (OECD 1982); longer retention times increase the sensitivity to nutrient loading.

There is a wide range of salinity and water quality among marine lakes depending on precipitation, connection to marine water, and the degree of stratification. pH is generally neutral to slightly acidic in dune lakes and neutral to slightly alkaline in rockland lakes (FNAI 2010). Similar to inland lakes, marine lakes are generally oligotrophic under undisturbed conditions, with low nutrient concentrations and low productivity. DO concentrations in the surface layer are often above existing criteria, while bottom DO concentrations can be low because of the decomposition of organic matter and limited or lacking reaeration. Their oligotrophic nature and stratification make marine lakes susceptible to the adverse effects of nutrient pollution.

The biologic composition of marine lakes depends on the salinity of the lake. Dense fringing wetland vegetation, similar to that found around freshwater lakes, is observed in many coastal dune lakes but not generally in rockland lakes (FNAI 2010). The freshwater layer of rockland lakes is an important refuge for freshwater invertebrates, and a variety of amphihaline invertebrates take advantage of those systems, although their diversity is typically smaller than that of water bodies with more moderate changes in salinity. Fish diversity is reduced relative to adjacent marine systems; the fauna is predominantly composed of mollies, sheepshead minnows, and mosquitofish. While higher trophic levels vary, a variety of reptiles and mammals rely on these systems, especially rockland lakes, which are important watering holes for Key Deer.

3.2.1. Definition and Classification

EPA proposes applying the FDEP definition of predominantly marine waters to classify lakes as marine lakes. Section 62-302.200 F.A.C. states that, "predominantly marine waters shall mean surface waters in which the chloride concentration at the surface is greater than or equal to 1,500 milligrams per liter." FDEP has already designated 12 Gulf Coast dune lakes west of St. Andrew Bay as Class III marine waters in state water quality standards (Figure 3-1). One of those lakes, Deer Lake, has a very limited connection to marine waters with an average salinity of 0.23 ppt and is the only marine lake that does not meet the 62-302.200 F.A.C. rule definition.

Figure 3-1. Locations of the 50 candidate marine lakes used in the assessment (yellow), including the 12 lakes designated in state water quality standards (red)

After analyzing available salinity data from Impaired Waters Rule (IWR) Run 40 and FDEP classifications, EPA has identified 50 candidate marine lakes for assessment, including the 12 lakes already designated in state standards (Figure 3-1). Additional supporting data are provided in Appendix F. Those 50 lakes were used to conduct the following analyses to support the proposed criteria for marine lakes, including the application of the inland criteria (USEPA 2010) to those marine lakes.

3.2.2. Water Quality

Water quality data were compiled from IWR Run 40 and FDEP's Integrated Water Resource Monitoring Network (see Appendix F) for the 50 candidate marine lakes. Summary statistics are presented in Table 3-1. The data were grouped into Class III marine (11 lakes), Class III freshwater (1 lake, Deer Lake), and additional (38 lakes) lake classes. The geometric mean was first calculated by parameter for each lake, and then the geometric mean was calculated for each lake category. The geometric standard deviation (GSD) is reported in Table 3-1. Geometric means were used for consistency with the proposed numeric nutrient criteria. Water quality data for individual lakes and box plots for selected parameters are provided in Appendix F.

Table 3-1. Geometric mean (± 1 GSD) values for water quality parameters derived from the geometric mean values for each lake; n = number of lakes. Confidence intervals were not possible for the single lake (Deer Lake) in the Class III freshwater category, however geometric mean values for Deer Lake are listed.

Parameter[a]	III Marine n=11 Geomean (1 GSD)	III Freshwater n=1 Geomean	Additional n=38 Geomean (1 GSD)
Alkalinity (mg/L as $CaCO_3$)	13.9 (5.8)	3.4	74.7 (2.6)
Chloride (mg/L)	1,388 (5.3)	20.9	2,844 (4.0)
Chl-a (µg/L)	3.1 (1.3)	5	5.3 (4.4)
Chl corrected (µg/L)	3.2 (2.7)	0.6	6.7 (4.0)
Color (PCU)	57.1 (2.1)	104	43.0 (3.9)
DO (mg/L)	5.1 (1.2)	6.3	5.4 (1.4)
TN (mg/L)	0.40 (1.4)	0.44	0.83 (1.7)
TP (mg/L)	0.013 (1.9)	0.011	0.038 (2.72)
Salinity (ppt)	3.2 (3.0)	0.1	5.0 (5.1)
Specific conductivity (µmhos/cm)	3,699 (4.5)	327.4	13,370 (2.7)
Temperature (°C)	22.1 (1.0)	20.8	24.3 (1.1)

$CaCO_3$=calcium carbonate; PCU=platinum-cobalt units; µmhos=micromhos

Geometric mean chloride, salinity, and conductivity values were all greater for the 38 additional marine lakes compared to the classified III marine lakes. Geometric mean nutrient and chl-a concentrations were also greater for the additional marine lakes indicating a higher potential for nutrient-related stress.

Among the 50 candidate marine lakes, episodes of hypoxia (DO < 2 mg/L) were relatively common, occurring in a few to more than 25 percent of observations depending on the individual lake (Appendix F, Figure F-10). Three of the lakes had more than 25 percent of observations less than 2 mg/L (lower quartile), and five lakes had median DO at or below 4 mg/L. The frequency and degree of stratification in the lakes is unknown.

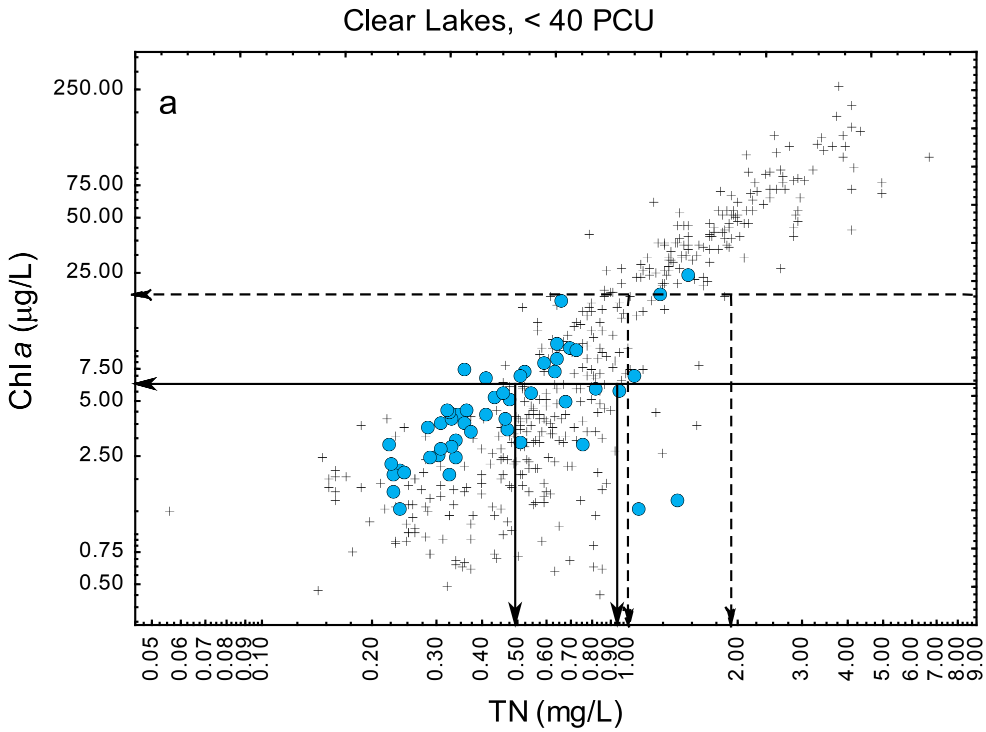
3.2.3. Response to Nutrients: Comparison of Inland and Marine Lakes

Thirty-three of the 50 marine lakes had sufficient data from IWR Run 40 and FDEP's Integrated Water Resource Monitoring Network (see Appendix F) to characterize color, TN, TP and chlorophyll response to nutrients. To compare to inland lakes that were used to derive the inland lake nutrient criteria, a minimum of four observations per year were required to calculate a lake-year geometric mean for TN, TP, and chl-a, respectively. Chl-a data for the marine lakes were primarily uncorrected chlorophyll, whereas the inland lakes data consisted of corrected chl-a only (USEPA 2010).

The marine lakes examined showed the same response relationship of chl-a to nutrients as the inland lakes (Figure 3-2 and Figure 3-3). The marine lakes are generally oligotrophic, and many have lower concentrations of TN and TP than their inland counterparts. The marine clear lakes fall on the chl-a–TN regression line similar to alkaline clear lakes (Figure 3-2a), but generally at lower TN concentrations. The marine clear lakes also appear to have slightly higher concentrations of TP for a given chl-a level (Figure 3-2b).

Only one marine lake, Alligator Lake, had high color (> 140 platinum cobalt units [PCU]), with a mean color of 172 PCU (two other marine lakes (Mud Bay and Salt Lake) had color in the range 140–200 PCU, but neither had TN data available). In general, lakes with color in the range 100–200 PCU are considered moderate in color, and are not significantly different within that range. Therefore the highly colored inland lakes (triangles) in Figure 3-3b are not applicable to marine lakes. Most marine lakes, especially those with color (> 40 PCU), had lower TN and TP concentrations compared to inland lakes (Figure 3-3), possibly because of their remoteness and lower anthropogenic inputs of nutrients.

EPA proposes that the inland lake criteria apply to Florida's marine lakes because they follow the trends and regressions of inland lakes.



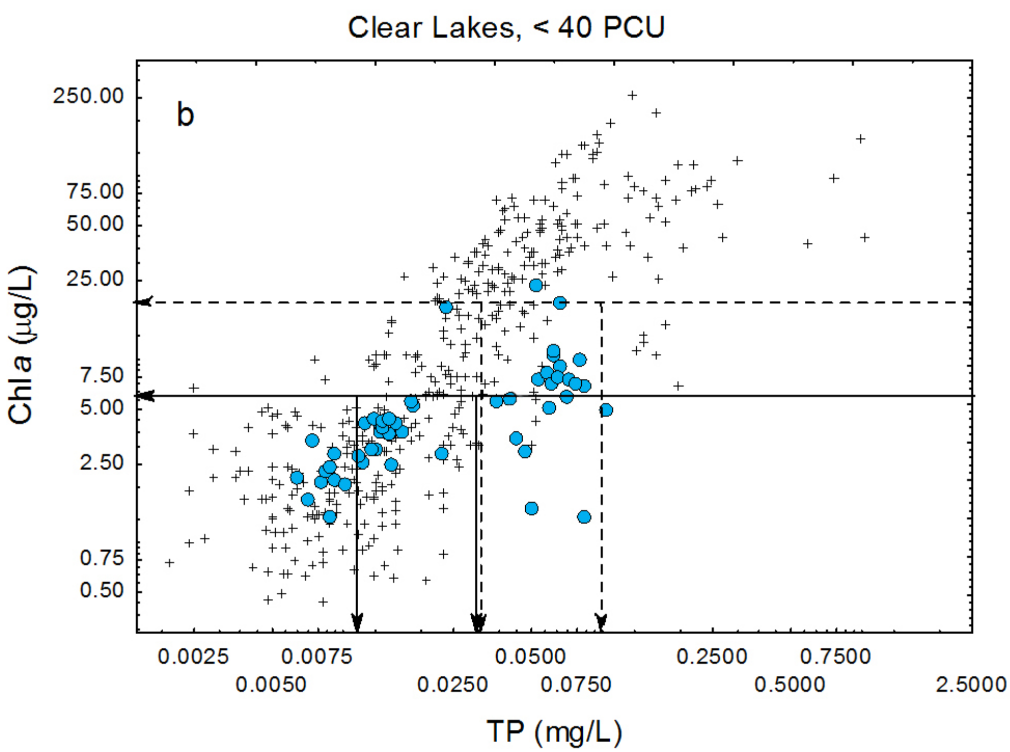


Figure 3-2. Chl-a–nutrient relationships for TN (a) and TP (b) for clear (< 40 PCU) marine lakes (N = 52 lakes years for 11 lakes) (filled circles), as compared to clear inland lakes (crosses). Horizontal arrows show inland chl-a criteria (solid: low alkalinity; dashed: high alkalinity), and vertical arrows show range of TN and TP inland criteria.

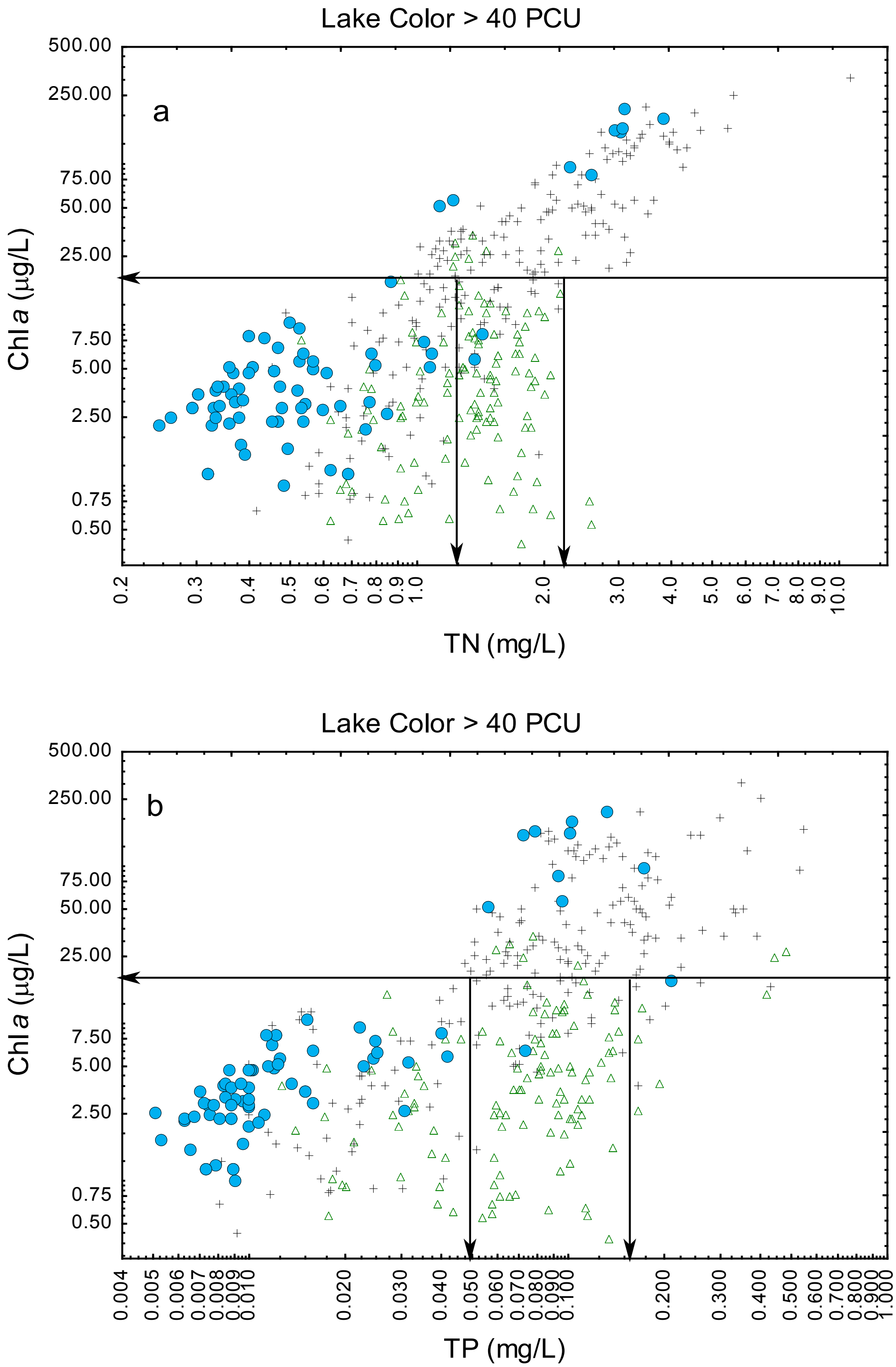


Figure 3-3. Chl-a–nutrient relationships for TN (a) and TP (b) for colored (> 40 PCU) marine lakes (N = 79 lake years for 22 lakes) (filled circles), as compared to inland lakes with moderate (40–140 PCU) (crosses) and high color (> 140 PCU) (triangles). Horizontal arrows show inland chl-a criteria (20 µg/L), and vertical arrows show range of TN and TP inland criteria.

3.2.4. Proposed Numeric Nutrient Criteria for Marine Lakes

EPA has determined that the inland lake criteria for TN, TP, and chl-a based on color and alkalinity are applicable to marine lakes because of similarities in trophic condition and chl-a response to nutrient concentrations (Figure 3-2 and Figure 3-3). Lakes also tend to have longer retention times than estuaries. Inland and marine lakes might contain different species (because of salinity) but they respond in similar ways to nutrient inputs and have similar biological endpoints related to nutrients and primary production (e.g., chl-a, DO). Therefore, the numeric criteria for freshwater lakes established in the inland rule are proposed for marine lakes.

The inland freshwater lake criteria to be applied to marine lakes as specified in *Water Quality Standards for the State of Florida's Lakes and Flowing Waters (Final Rule)* (USEPA 2010) are shown in Table 3-2.

Table 3-2. EPA's numeric nutrient criteria derived for inland freshwater lakes (USEPA 2010) and applied to marine lakes

Lake Color[a] and Alkalinity	Chl-a[b,*] mg/L	TN mg/L	TP mg/L
Colored lakes[c]	0.020	1.27 [1.27-2.23]	0.05 [0.05-0.16]
Clear lakes, high alkalinity[d]	0.020	1.05 [1.05-1.91]	0.03 [0.03-0.09]
Clear lakes, low alkalinity[e]	0.006	0.51 [0.51-0.93]	0.01 [0.01-0.03]

[a] PCU assessed as true color free from turbidity
[b] Chl-a is defined as corrected chlorophyll, or the concentration of chl-a remaining after the chlorophyll degradation product, phaeophytin a, has been subtracted from the uncorrected chl-a measurement
[c] Long-term color > 40 PCU and alkalinity > 20 mg/L $CaCO_3$
[d] Long-term color ≤ 40 PCU and alkalinity > 20 mg/L $CaCO_3$
[e] Long-term color ≤ 40 PCU and alkalinity ≤ 20 mg/L $CaCO_3$
* For a water body, the annual geometric mean of chl-a, TN or TP concentrations shall not exceed the applicable criterion concentration more than once in a three-year period

3.2.5. Application of the Inland Lake Criteria to 50 Marine Lakes

The 50 candidate marine lakes were compared to the inland freshwater lake criteria. Inland criteria for freshwater lakes are expressed as an annual geometric mean of TN, TP, and chl-a, not to be exceeded more than once in a 3-year period. Annual geometric means for each parameter were calculated for each marine lake-year and compared to the applicable inland criteria. Alkalinity and color data were used to classify each marine lake into clear/colored and high/low alkalinity classes using the criteria used for inland lakes. Existing monitoring data were used to assess the degree to which marine lakes met the inland criteria using three categories: met the criteria, within the range, or exceeded the criteria. Additional details are provided in Appendix F.

There were sufficient monitoring data from IWR Run 40 and FDEP's Integrated Water Resource Monitoring Network (see Appendix F) to assess 41 of the 50 marine lakes (Table 3-3). Of the 41 lakes with data, twelve lakes could not be classified for color and alkalinity. Of the remaining 29 lakes, 23 (79%) were colored lakes, 6 (21%) were clear lakes with high alkalinity, and no lakes were clear lakes with low alkalinity. Of the 12 lakes identified by FDEP as Class III marine waters, 11 either meet the freshwater lake criteria or are in range of the freshwater lake criteria. Of 29 lakes identified by EPA as marine lakes with sufficient nutrient data, 13 either meet the freshwater lake criteria or are in range of the freshwater lake criteria, while the remaining 16 exceed the numerical value of the freshwater lake criteria. However, of the 16 lakes that exceed the inland criteria, 9 have TN, TP, and chl-a concentrations within the range provided for potentially receiving modified criteria, under the modified criteria provisions of the *Final Rule.* It is acknowledged that of the 16 lakes that were identified as exceeding the numerical value of the freshwater lake criteria, 10 had only one year of monitoring data available, which was insufficient to evaluate the duration and frequency component of the inland *Final Rule.*

Table 3-3. Number of lakes that meet, exceed, or are in range of the inland freshwater lake criteria; in parentheses are the number of lakes where chl-a, TN, and TP met the test for receiving potentially modified nitrogen and phosphorus criteria

	Lake Class														
	III Marine					III Fresh					Additional				
Color and Alkalinity	Number	Meets	In range	Exceeds	Exceeds[a]	Number	Meets	In range	Exceeds	Exceeds[a]	Number	Meets	In range	Exceeds	Exceeds[a]
Colored lakes	6	6	-	-	-	1	1	-	-	-	16	6	-	3	7 (3)
Clear lakes, high alkalinity	1	1	-	-	-	-	-	-	-	-	5	2	-	2 (1)	1 (1)
Clear lakes, low alkalinity	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
No data	4	-	3	1 (1)	-	-	-	-	-	-	8	2	3	1 (1)	2 (2)
Totals	**11**	**7**	**3**	**1 (1)**	**0**	**1**	**1**	**0**	**0**	**0**	**29**	**10**	**3**	**6 (2)**	**10 (6)**

[a] Number of lakes with only 1 year of data, no assessment of duration and frequency

3.3. References

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