NATIONAL MARINE FISHERIES SERVICE
ENDANGERED SPECIES ACT SECTION 7
BIOLOGICAL OPINION

Title:	Biological Opinion on the Approval of Florida Estuary Specific Numeric Nutrient Criteria under section 303(c) of the Clean Water Act for Total Phosphorus, Total Nitrogen, and Chlorophyll-a in 42 Estuary Segments
Consultation Conducted By:	Endangered Species Act Interagency Cooperation Division, Office of Protected Resources, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, U.S. Department of Commerce
Action Agency:	Environmental Protection Agency, Region 4, Atlanta, Georgia
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UNITED STATES DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
NATIONAL MARINE FISHERIES SERVICE
Silver Spring, MD 20910

AUG 16 2017

Refer to NMFS No.: FPR-2017-9187

Memorandum For: Joanne Benante
Chief, Water Quality Planning Branch
U. S. Environmental Protection Agency, Region 4

From: Cathryn E. Tortorici
Chief, Endangered Species Act Interagency Cooperation Division

Subject: Biological Opinion on the Environmental Protection Agency (EPA) Approval of Florida Estuary Specific Numeric Nutrient Criteria under section 303(c) of the Clean Water Act for Total Phosphorus, Total Nitrogen, and Chlorophyll-a in 42 Estuary Segments Pursuant to Section 7 of the Endangered Species Act of 1973

Enclosed is the National Marine Fisheries Service's (NMFS) biological opinion on the effects of the EPA's approval of Florida estuary specific numeric nutrient criteria under section 303(c) of the Clean Water Act for total phosphorus, total nitrogen, and chlorophyll-a in 42 estuary segments on endangered and threatened species under NMFS's jurisdiction and critical habitat that has been designated for those species. We have prepared the biological opinion pursuant to section 7(a)(2) of the Endangered Species Act, as amended (ESA; 16 U.S.C. 1536(a)(2)).

Based on our assessment, we concluded that EPA's approval of these standards is not likely to jeopardize any ESA-listed species under NMFS' jurisdiction. We also conclude that EPA's approval of these standards is not likely to destroy or adversely modify designated critical habitat under NMFS' jurisdiction.

We based our conclusion on the performance of the numeric nutrient criteria in preventing elevated chlorophyll-a levels indicative of eutrophic conditions. To arrive at our conclusion, these analyses were integrated with the current status of ESA-listed species under NMFS' jurisdiction, baseline conditions in Florida waters where these species and their designated critical habitat occur, and the cumulative effects of future State or private activities that are reasonably certain to occur within the action area.

This concludes formal consultation on this action. Consultation on this issue must be reinitiated if: (1) the amount or extent of allowable take is exceeded for the identified action; (2) new information reveals effects of this action that may affect listed species or critical habitat in a manner or to an extent not previously considered; (3) the identified action is subsequently modified in a manner that causes an effect to the listed species that was not considered in this consultation; or (4) a new species is listed or critical habitat designated that may be affected by the action.

If you have any questions regarding this biological opinion, please contact Pat Shaw-Allen, consultation biologist, at (301) 427-8473, or by e-mail at pat.shaw-allen@noaa.gov, or myself at (301) 427-8495 or by e-mail at cathy.tortorici@noaa.gov.

WEDNESDAY, AUGUST 16, 2017

NATIONAL MARINE FISHERIES SERVICE
ENDANGERED SPECIES ACT SECTION 7 BIOLOGICAL OPINION

Action Agency: Environmental Protection Agency,
Region 4, Atlanta, Georgia

Activity Considered: Approval of Florida Estuary Specific Numeric Nutrient Criteria under section 303(c) of the Clean Water Act for Total Phosphorus, Total Nitrogen, and Chlorophyll-a in 42 Estuary Segments

Consultation Conducted By: Endangered Species Act Interagency Cooperation Division, Office of Protected Resources, National Marine Fisheries Service

Approved:

For Donna S. Wieting
Director, Office of Protected Resources

Date: 8/16/17

Public Consultation Tracking System (PCTS) number: FPR-2017-9187

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1 INTRODUCTION

Section 7(a)(2) of the Endangered Species Act of 1973, as amended (16 USC. 1531 et seq.; ESA) requires Federal agencies to insure that their actions are not likely to jeopardize the continued existence of endangered or threatened species or adversely modify or destroy their designated critical habitat. The definition of species in the Endangered. Species Act "… includes any subspecies of fish or wildlife or plants, or any distinct population segment (DPS)." When a Federal agency's action "may affect" a protected species, that agency is required to consult with The National Oceanic and Atmospheric Administration (NOAA) National Marine Fisheries Service (NMFS) or the U.S. Fish and Wildlife Service (USFWS), depending upon the endangered species, threatened species, or designated critical habitat that may be affected by the action (50 CFR §402.14 (a)). Federal agencies must engage in formal consultation if their actions are likely to adversely affect listed species or designated critical habitat. If Federal agencies have concluded that an action "may affect, but is not likely to adversely affect" (i.e., NLAA) endangered species, threatened species, or designated critical habitat, they may alternatively engage in "informal consultation" and request NMFS and/or USFWS concurrence with that conclusion (50 CFR §402.14(b)).

Section 7(b)(3) of the ESA requires that at the conclusion of formal consultation, NMFS and/or USFWS provide a biological opinion stating how the Federal agencies' actions will affect ESA-listed species and their designated critical habitat under their jurisdiction. If any incidental take is expected, section 7(b)(4) requires the consulting agency to provide an incidental take statement (ITS) that specifies the impact of any incidental taking and includes reasonable and prudent measures (RPMs) to minimize such impacts.

In this case, the action is the Environmental Protection Agency's (EPA) approval of Numeric Nutrient Criteria (NNC) applied by the state of Florida (hereinafter referred to as the "State" or "Florida") in its 305(b)/303(d) water quality assessment program to identify waters currently in or approaching impaired conditions.

NMFS prepared this biological opinion (opinion) and ITS in accordance with section 7(b) of the ESA and implementing regulations at 50 CFR §402. This document represents NMFS' final opinion on the effects of these actions on endangered and threatened species and designated critical habitat that has been designated for those species.

In this opinion, we evaluate whether the EPA approval of criteria for nutrient levels in specific estuaries that are waters of the U.S. is likely to jeopardize endangered and threatened species or destroy or adversely modify designated critical habitat. An endangered species is defined by the ESA as a species in danger of extinction throughout all or a significant portion of its range; a threatened species is defined as a species likely to become an endangered species throughout all or a significant portion of its range in the foreseeable future. The continued existence of a population is determined by the fate of the individuals within it and the continued existence of a species is determined by the fate of its populations. Populations grow or decline as its individuals live, die, grow, mature, migrate, and reproduce, or fail to do so. Designated critical habitat is defined as the specific areas within the geographical area occupied by the species, at the time it is listed, on which are found those physical or biological features that are essential to the conservation of the species, and which may require special management considerations or protection. Designated critical habitat can also include specific areas outside the geographical area occupied by the species at the time it is listed that are determined by the Secretary to be essential for the conservation of the species[1]. Destruction or adverse modification means a direct or indirect alteration that appreciably diminishes the value of designated critical habitat for the conservation of a listed species. Such alterations may include, but are not limited to, those that alter the physical or biological features essential to the conservation of a species or that preclude or significantly delay development of such features. These include, but are not limited to:

- space for individual and population growth, and for normal behavior;
- nutritional or physiological requirements (e.g., food, water, air, light, minerals);
- cover or shelter;
- sites for breeding, reproduction, rearing of offspring, germination, or seed dispersal; and
- habitats that are protected from disturbance or are representative of the historic, geographic, and ecological distributions of a species.

This opinion, and ITS, were prepared by the ESA Interagency Cooperation Division in accordance with section 7(b) of the ESA and implementing regulations at 50 CFR §402. This opinion complies with the Data Quality Act (44 USC. 3504(d)(1) et seq.) and underwent pre-dissemination review. This document represents NMFS' opinion on the effects of these actions on endangered and threatened species and designated critical habitat that has been designated for those species. A complete record of this consultation is on file at NMFS' Office of Protected Resources in Silver Spring, Maryland.

1.1 Background

The Florida Wildlife Federation filed a lawsuit in 2008 seeking to require EPA to promulgate water quality criteria for nutrients in Florida waters. On January 14, 2009, the EPA determined that numeric nutrient water quality criteria in Florida were necessary to meet the requirements of the Clean Water Act (CWA). In August 2009, the EPA entered into a Consent Decree with the Florida Wildlife Federation to settle the 2008 litigation, setting dates for promulgation of NNC. The consent decree provided that, if Florida submits and the EPA approves State NNC for the relevant waters before any of the above dates, the EPA would no longer be under obligation with respect to promulgating such criteria.

On June 13, 2012, the Florida Department of Environmental Protection (Florida) submitted new and revised NNC for review by the EPA pursuant to section 303(c) of the CWA. On May 22,

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[1] Sections 3 (5), (6) and (20), 16 USC. 1532 (5), (6) and (20)

2015, Florida submitted final estuary NNC for the EPA's review and approval under Section 303(c) of the CWA. On August 2, 2016 NMFS had completed its review of EPA's approval of estuary NNC for those NNC established in 2012 that had not been changed in the final 2015 NNC. NMFS biological opinion concluded the EPA's approval of those revisions were not likely to jeopardize any ESA-listed species and were not likely to destroy or adversely modify designated critical habitat. NMFS could not assess that subset of NNC that changed in the 2015 submission because EPA had not yet arrived at a determination of "no effect," "may affect/not likely to adversely affect" ESA-listed species or designated critical habitat (NLAA) or "may affect/likely to adversely affect" ESA-listed species or designated critical habitat (LAA).

1.2 Consultation History

- On February 3, 2017, EPA Region 4 transmitted a letter to the NMFS' Silver Spring Office requesting reinitiation of consultation and concurrence on their determination that Florida's 2015-revised estuary NNC are NLAA. The letter also indicated that, in addition to supplemental materials supplied with the letter, its biological evaluation requirement is fulfilled by incorporating the original Florida 2012 NNC biological evaluation by reference because the action is identical or very similar to a previous action in that evaluation, pursuant to the ESA regulations at 50 C.F.R. §402.12(g).
- On March 13, 2017, NMFS responded to EPA's letter explaining that, because the revised criteria have not yet been evaluated, this consultation is technically not a reinitiation; rather it is a new consultation. NMFS agreed that the information required to proceed with consultation was already provided by EPA or was otherwise publically available. However, NMFS could not concur with EPA's NLAA determination because preliminary analyses of available monitoring data indicated that the proposed total nitrogen (TN) and total phosphorous (TP) NNC for the Loxahatchee Estuary were associated with Chlorophyll-a-(Chl-a) concentrations exceeding Chl-a criteria, suggesting these estuary NNC would actually promote eutrophication in the estuary.
- On March 29, 2017, EPA transmitted a copy of the "Loxahatchee River Water Quality Monitoring Task 2: Final Report Assessment of 2014-2015 Loxahatchee River Water Quality" (Stoner et al. 2016)
- On July 6, 2017, EPA and NMFS agreed to extend the deadline for a signed opinion to August 18, 2017.

2 THE ASSESSMENT FRAMEWORK

Section 7(a)(2) of the ESA requires Federal agencies, in consultation with NMFS, to ensure that their actions are not likely to jeopardize the continued existence of endangered or threatened species; or adversely modify or destroy their designated critical habitat.

"Jeopardize the continued existence of" means to engage in an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of an ESA-listed species in the wild by reducing the reproduction, numbers, or distribution of that species." 50 CFR 402.02.

"Destruction or adverse modification" means a direct or indirect alteration that appreciably diminishes the value of designated critical habitat for the conservation of an ESA-listed species. Such alterations may include, but are not limited to, those that alter the physical or biological features essential to the conservation of a species or that preclude or significantly delay development of such features (50 CFR 402.02). An ESA section 7 assessment involves the following steps:

Description of the Proposed Action (Section 3), Interrelated and Interdependent Actions (Section 4), and Action Area (Section 5): We describe the proposed action and those aspects (or stressors) of the proposed action that may have direct or indirect effects on the physical, chemical, and biotic environment, we identify any interrelated and interdependent actions, and describe the action area with the spatial extent of those stressors.

Status of Species and Designated Critical Habitat (Section 6): We identify the ESA-listed species and designated critical habitat that are likely to co-occur with those stressors in space and time and evaluate the status of those species and habitat.

Environmental Baseline (Section 7): We describe the environmental baseline in the action area including: past and present impacts of Federal, state, or private actions and other human activities in the action area; anticipated impacts of proposed Federal projects that have already undergone formal or early section 7 consultation, impacts of state or private actions that are contemporaneous with the consultation in process.

Effects of the Action (Section 8): *Problem Formulation (*Section 8.1): To determine the effects of the action, we conduct a problem formulation that first identifies the stressors of the action and their potential effects. In the subsequent *Risk Analysis*, we evaluate the potential adverse effects of Florida's criteria on ESA-listed species and designated critical habitat under NMFS' jurisdiction. To do this, we begin with problem formulation that integrates the stressors of the action and their potential effects with the species status (Section 6) and the Environmental Baseline (Section 7) and formulate risk hypotheses. The risk hypotheses identify assessment endpoints of concern for ESA-listed species and designated critical habitat. To evaluate the risk hypotheses, we consider the potential exposure of individual members of ESA-listed species (exposure analysis, Section 8.2) and essential features of designated critical habitat, and what expected responses might be (response analysis, Section 8.2.2). If the assessment endpoints of the individuals or the essential features indicate adverse effects, we evaluate whether those responses would affect populations or subpopulations of species or the designated critical habitat (risk characterization).

Cumulative Effects (Section 9): Cumulative effects are the effects to ESA-listed species and designated critical habitat of future state or private activities that are reasonably certain to occur within the action area. 50 CFR 402.02. Effects from future Federal actions that are unrelated to the proposed action are not considered because they require separate ESA section 7 compliance.

Integration and Synthesis (Section 10): In this section, we integrate the analyses in the opinion to summarize the consequences to ESA-listed species and designated critical habitat under NMFS' jurisdiction.

Conclusion (Section 11): With full consideration of the status of the species and the designated critical habitat, we consider the effects of the action within the action area on populations or subpopulations and on essential habitat features when added to the environmental baseline and the cumulative effects to determine whether the action could reasonably be expected to:

- Reduce appreciably the likelihood of survival and recovery of ESA-listed species in the wild by reducing its numbers, reproduction, or distribution, and state our conclusion as to whether the action is likely to jeopardize the continued existence of such species; or
- Appreciably diminish the value of designated critical habitat for the conservation of an ESA-listed species, and state our conclusion as to whether the action is likely to destroy or adversely modify designated critical habitat.

If, in completing the last step in the analysis, we determine that the action under consultation is likely to jeopardize the continued existence of ESA-listed species or destroy or adversely modify designated critical habitat, then we must identify RPA(s) to the action, if any, or indicate that to the best of our knowledge there are no RPAs. See 50 C.F.R. § 402.14.

In addition, we include an ITS that specifies the impact of the take, RPMs to minimize the impact of the take, and Terms and Conditions to implement the RPMs. ESA section 7 (b)(4); 50 CFR 402.14 (i). We also provide discretionary conservation recommendations that may be implemented by EPA. 50 CFR 402.14 (j). Finally, we identify the circumstances in which reinitiation of consultation is required. 50 CFR 402.16.

To comply with our obligation to use the best scientific and commercial data available, we collected information identified through searches of Web of Science, scientific publisher databases (e.g., Elsevier), government databases (e.g., EPA's National Service Center for Environmental Publications), and literature cited sections of peer reviewed articles, species listing documentation, and reports published by government and private entities. This opinion is based on our review and analysis of various information sources, including:

- EPA' s biological evaluations for Florida's estuary nutrient criteria;

- fact sheets and technical support documents;
- data from EPA's STOrage and RETrieval database (STORET);
- the August 2, 2016 opinion on EPA's approval of Florida’s NNC;
- status reviews, recovery plans, and listing notices for ESA-listed species and designated critical habitat;
- reports on the status and trends of water quality; and
- the best available commercial and scientific information, including peer reviewed research.

These resources were used to identify information relevant to the potential stressors and responses of ESA-listed species and designated critical habitat under NMFS’ jurisdiction that may be affected by the proposed action to draw conclusions on risks the action may pose to the continued existence of these species and the value of designated critical habitat for the conservation of ESA-listed species.

3 DESCRIPTION OF THE ACTION

Under Section 7(a)(2) of the ESA “action” means all activities or programs of any kind authorized, funded, or carried out, in whole or in part, by federal agencies in the United States or upon the high seas. The action agency for this consultation is Region 4 of the EPA. The action is EPA Region 4’s potential approval of revisions to Florida’s water quality standards (WQS), including numeric nutrient criteria for certain estuaries, contained in Rule 62-302 of the Florida Administrative Code. To ensure that the opinion is consistent with the agency’s action, the following *Description of the Action* was reviewed in draft form by EPA Region 4 and revised according to their edits. EPA was notified of, and concurred with, necessary edits made to the description of the action during the development of this opinion.

The EPA’s statutory authority for the review and approval of state WQS falls under Section 303(c) of the CWA, which establishes the basic structure for regulating discharges of pollutants into and regulating quality standards for the waters of the United States. Under section 303(c), states are required to adopt WQS to restore and maintain the chemical, physical, and biological integrity of the nation’s waters. EPA is required to review these changes to ensure revisions in designated water uses are consistent with the CWA and that new or revised standards protect the designated water uses. Approval must be granted by EPA within 60 days or EPA must respond within 90 days if it disapproves of the standards. Specifically the federal WQS regulations at 40 CFR § l31.21 state, in part, that when EPA disapproves a state's WQS, EPA shall specify changes that are needed to ensure compliance with the requirements of Section 303(c) of the CWA and federal WQS. The adoption, review, and approval of state WQS satisfy the goals and policies of the CWA (33 USC §§1251).

On February 3, 2017, EPA requested the re-initiation of consultation for concurrence on the above mentioned criteria revisions. Based on the previous work completed by the EPA, the letter provided historical evaluations as well as providing some new, supplemental information for NMFS to consider in this latest consultation effort. On March 13, 2017, NMFS clarified that the activity would be part of a new consultation, given that some of the criteria which were revised, were not previously consulted on with NMFS. With the acceptance of the previous materials to be incorporated by reference, the NMFS response letter indicated the Agency had received the information needed to complete its analysis and likely non-concurrence with certain revisions. Following a March 23, 2017 phone conversation between Pat Shaw-Allen, NMFS, and Lauren Petter, EPA, it was confirmed that the 135-day period for a response started from the date of the NMFS response letter to EPA. Additional estuary information was provided by email in March based on a follow up question from Pat Shaw-Allen and information provided by Florida in response to that inquiry.

3.1 Environmental Protection Agency Determinations on the Revised Estuary Numeric Nutrient Criteria for Total Nitrogen, Total Phosphorous, and Chlorophyll–a

EPA’s analysis and conclusions on Florida's derivation of NNCs are outlined in the BE documents for the prior consultation, associated decision documents, and guidance. These are summarized in the original August 2, 2016 opinion for the consultation on EPA's approval of Florida's 2012 and 2013 NNC submissions. Additional analyses provided in the BE for the recently revised NNC which are the subject of the current consultation supported EPA's NLAA determinations through describing how and why the criteria were changed and qualitative evaluations of the scientific defensibility of Florida's methodology in revising the criteria.

In many cases, NNC were revised based on changes in the size of the underlying dataset, and the resulting statistic that was selected to derive the final revised criterion value. Some changes were related to a shift in approach, such as moving from a modeled approach to a distributional statistics approach, change in model used, or consideration of the amount of landscape development in a given region (i.e., a broader prediction interval was allowed for more natural, less developed watersheds). These revisions resulted in a change in the magnitude, statistic, and sometimes duration and frequency of a criterion. The EPA determined that the criteria revisions are either identical in approach, reflect minor modifications to past acceptable approaches, or represent a scientifically defensible approach that differs from what was used previously.

3.2 Derivation of Estuary-Specific Criteria: Healthy Conditions and Total Maximum Daily Loads

Each Florida estuarine system is subdivided into segments based on physical factors and long-term average salinity gradients before evaluating each segment to determine whether the current conditions were protecting the most sensitive designated uses. Most estuary standards are based on distributional statistics applied to data for reference conditions. However, Total Maximum Daily Loads (TMDLs) were submitted as site-specific standards for those segments that were currently or had been previously identified on the state's 303(d) impaired waters list as impaired for nutrients or dissolved oxygen (DO).

Florida's estuary-specific NNC, including those considered in this opinion, were derived through different strategies determined by the quantity and comprehensiveness of available data, and, in some cases, the complexity of the system to be protected. Consequently, the NNC are expressed in different terms. Most are expressed as annual geometric mean (AGM) not to be exceeded more than once in three years or as a single sample value not to be exceeded more than ten percent of the time (PCT), and a few are based on long term 7-year averages (LTAVG). Some are expressed as maximum annual loads, and others are expressed as daily loads. In the action under consultation, EPA has only approved revisions that resulted in a less stringent criterion or in different expressions of duration or frequency (e.g., AGM versus 10 percent of samples), so only such revisions are addressed in this opinion.

3.2.1 Florida’s Reference Condition Approach for Numeric Nutrient Criteria Approved by the Environmental Protection Agency

Florida applied a reference condition approach using distributional statistics to most estuaries. Distributional statistics are used to set NNC at a level that will maintain the current distribution of reference condition monitoring data, while accounting for natural temporal variability. Reference conditions were based on either reference period or reference site data. Reference period data are selected from a period when the water itself was determined to be biologically unimpaired and supporting its most sensitive designated uses. When reference period data were not available, data from an unimpaired, adjacent, and functionally similar reference site, a reference system, were used to represent reference conditions. Eight years of data were available for the derivation of standards for most estuary segments. For each of these segments, the annual geometric mean standard not to be exceeded more than once in a three-year period is based on the 80th percent prediction limit of average annual geometric means (i.e., 80 percent confidence that a new observation greater than the NNC does not reflect reference conditions). For those segments with less than eight years of data, but having at least 30 total samples, a "single-sample standard" not to be exceeded in more than 10 percent of samples was determined to be the upper 90th percent prediction limit of the samples. These concentration-based estuary NNC are open water, area-wide averages.

The reference condition approach was applied to some segments which had been previously identified as impaired by nutrients or DO, but had since attained designated uses. In these cases, data from years when the segment was impaired or from specific impaired portions of the segment were excluded from the calculation. The Florida determined reference conditions using biological endpoint data from currently unimpaired segments or data from the period of time when a segment was unimpaired. Endpoints used in this determination included DO concentration and/or percent saturation, chlorophyll-a concentration, and the seagrass indicators: colonization depth, water clarity, coverage, and extent. Specifically, achievement of 20 percent of the surface light at the bottom of the water column is considered protective of seagrass communities and a chlorophyll-a concentration of 20 µg/L, not to be exceeded more than 10 percent of the time, is considered indicative of balanced algal populations. Taken with spatial attributes of seagrass indicators and Florida's DO criteria previously approved by EPA and applied by Florida, EPA concluded that these endpoints are expected to indicate the health of the system as a whole and, at reference levels, represent conditions that protect aquatic life and recreation uses.

EPA determined that the reference condition approach applied reliable, vetted, and representative data in calculating is estuarine standards. The EPA concluded that because the established NNC are associated with nutrient levels that are necessary to protect designated uses from harmful nutrient concentrations, the NNC approved by the EPA are NLAA for ESA-listed species or their designated critical habitat. The adoption of these standards and the EPA's subsequent approval of them provide numeric levels that can be used in assessment and permitting.

3.2.2 Florida's Mechanistic Modeling Approach for Numeric Nutrient Criteria Approved by EPA

Mechanistic modeling was applied when available data and/or the existing conditions were not suitable for distributional statistics. The effort applied peer reviewed models to estimate the quantity of water and pollutants associated with runoff from rain events associated with the contributing watershed of the estuary. Where data were available, a hydrodynamic model was linked to the Water Quality Analysis Simulation Program to simulate eutrophication-rates and effects. However, in Florida's Big Bend region, it was necessary to base watershed loadings on the most recent land cover information, simulated for the 1997-2009 period and the hydrodynamic and water quality modeling for the 2002-2009 period. This method was adapted for those waterbodies with low land use intensity and naturally low DO.

3.3 Estuary Segment-specific Criteria Addressed in this Opinion

The estuary segment-specific NNC derived under these differing strategies for the waters addressed in this action are listed in Table 1 below.

Table 1. Estuary Segments and Nutrient Criteria Addressed in this Opinion.

Estuaries and Segments	Chl-a Target	TN Criteria	TP Criteria	Criteria derived using...
Apalachicola Bay and Alligator Harbor				
Alligator Harbor	6.0 µg/L (AGM)	0.42 mg/L (AGM)	0.028 mg/L (AGM)	reference period
Apalachicola Offshore	8.2 µg/L (PCT)	0.57 mg/L (PCT)	0.032 mg/L (PCT)	reference system
Big Bend and Apalachee Bay				
Econfina River Estuary	4.9 µg/L (AGM)	1.14 mg/L (AGM)	0.101 mg/L (AGM)	hydrodynamic model
Fenholloway River Estuary	4.6 µg/L (AGM)	LOAD	LOAD	hydrodynamic model
Aucilla Offshore	9.5 µg/L (PCT)	0.60 mg/L (PCT)	0.025 mg/L (PCT)	reference period
Aucilla River Estuary	2.2 µg/L (PCT)	0.89 mg/L (PCT)	0.080 mg/L (PCT)	reference period
Ochlockonee River Estuary	9.2 µg/L (PCT)	0.86 mg/L (PCT)	0.067 mg/L (PCT)	reference period
Ochlockonee/Alligator Harbor Offshore	8.2 µg/L (PCT)	0.57 mg/L (PCT)	0.032 mg/L (PCT)	reference period
Spring Warrior Offshore	8.3 µg/L (PCT)	0.67 mg/L (PCT)	0.047 mg/L (PCT)	reference period
St. Marks Offshore	8.0 µg/L (PCT)	0.63 mg/L (PCT)	0.045 mg/L (PCT)	reference period
St. Marks River Estuary	6.0 µg/L (PCT)	0.70 mg/L (PCT)	0.044 mg/L (PCT)	reference period
Steinhatchee River Estuary	3.9 µg/L (AGM)	0.86 mg/L (AGM)	0.062 mg/L (AGM)	reference period
Charlotte Harbor/Estero Bay				
Moorings Bay	8.1 µg/L (AGM)	0.85 mg/L (PCT)	0.04 mg/L (PCT)	reference period
Guana River/Tolomato River/Matanzas River Estuary				
Pellicer Creek Estuary	4.3 µg/L (AGM)	1.10 mg/L (AGM)	0.123 mg/L (AGM)	reference period
Halifax and Tomoka River Estuaries				
Tomoka River Estuary	7.2 µg/L (AGM)	1.24 mg/L (AGM)	0.132 mg/L (AGM)	reference system
Indian River Lagoon System				
Banana River Lagoon	7.3 µg/L (AGM)	LOAD	LOAD	TMDL (Chl-a reference period)
Central Indian River Lagoon	5.9 µg/L (AGM)	LOAD	LOAD	TMDL (Chl-a reference period)
Newfound Harbor	7.3 µg/L (AGM)	LOAD	LOAD	TMDL (Chl-a reference period)
North Indian River Lagoon	6.4 µg/L (AGM)	LOAD	LOAD	TMDL (Chl-a reference period)
from St. Lucie Estuary to Ft. Pierce	4.7 µg/L (AGM)	0.72 mg/L (AGM)	0.070 mg/L (AGM)	reference period
between Hobe Sound and St. Lucie	6.9 µg/L (PCT)	0.63 mg/L (AGM)	0.060 mg/L (AGM)	reference system

Estuaries and Segments	Chl-a Target	TN Criteria	TP Criteria	Criteria derived using...
Ft. Pierce Inlet to Indian River County Line	4.7 µg/L (AGM)	0.72 mg/L (AGM)	0.070 mg/L (AGM)	reference system
Sebastian River Estuary	5.9 µg/L (AGM)	LOAD	LOAD	reference system
Sykes Creek Estuary	7.3 µg/L (AGM)	LOAD	LOAD	reference system
Intracoastal Waterway (ICWW)				
ICWW Palm Coast	4.5 µg/L (AAM)	LOAD	LOAD	TMDL
Palm Beach County ICWW	13.4 µg/L (PCT)	1.17 mg/L (PCT)	0.146 mg/L (PCT)	reference period
ICWW between North Lake Worth Lagoon and Lower Loxahatchee River	4.7 µg/L (AGM)	0.66 mg/L (AGM)	0.035 mg/L (AGM)	reference period
Lower St. Johns River and Tributaries				
Lower St. Johns River and Tributaries	5.4 µg/L (LTAVG)	LOAD	LOAD	TMDL (modeled for chl-a)
Loxahatchee River Estuary				
Loxahatchee River (Southwest Fork)	5.5 µg/L (AGM)	1.26 mg/L (AGM)	0.075 mg/L (AGM)	reference system
Mosquito River Lagoon				
Edgewater to Oak Hill	3.4 µg/L (AGM)	0.65 mg/L (AGM)	0.048 mg/L (AGM)	reference period
Oak Hill to Southern Terminus	2.5 µg/L (AGM)	1.14 mg/L (AGM)	0.034 mg/L (AGM)	reference period
Ponce de Leon to Edgewater	4.0 µg/L (AGM)	0.51 mg/L (AGM)	0.049 mg/L (AGM)	reference period
Springs Coast (Crystal River to Anclote River)				
Kings Bay	5.7 µg/L (AGM)	LOAD	LOAD	TMDL (Chl-a reference period)
St. Lucie Estuary				
Upper North Fork St. Lucie River	6.7 µg/L (AGM)	LOAD	LOAD	TMDL (Chl-a reference period)
Upper South Fork St. Lucie River	5.0 µg/L (AGM)	LOAD	LOAD	TMDL (Chl-a reference period)
Lower North Fork St. Lucie River	7.4 µg/L (AGM)	LOAD	LOAD	TMDL (Chl-a reference system)
Lower South Fork St. Lucie River	6.7 µg/L (AGM)	LOAD	LOAD	TMDL (Chl-a reference system)
St. Lucie Estuary	5.9 µg/L (AGM)	LOAD	LOAD	TMDL (Chl-a reference system)
Manatee Creek	5.9 µg/L (AGM)	0.72 mg/L (LTAVG)	0.081 mg/L (LTAVG)	reference system
St. Marys River				
Lower St. Marys River	12.9 µg/L (AGM)	0.77 mg/L (AGM)	0.181 mg/L (AGM)	reference period
Middle St. Marys River	8.0 µg/L (PCT)	1.12 mg/L (AGM)	0.113 mg/L (AGM)	reference period
Upper St. Marys River	3.0 µg/L (PCT)	1.35 mg/L (AGM)	0.093 mg/L (PCT)	reference period

4 INTERRELATED AND INTERDEPENDENT ACTIONS

The NMFS has not identified any additional interdependent or interrelated actions for EPA's approval of NNC for the estuary segments addressed in this opinion.

5 ACTION AREA

The action area is defined as all areas to be affected directly or indirectly by the federal action and not merely the immediate area involved in the action (50 CFR §402.02). For this opinion, the action area includes specific estuary segments within the state of Florida to which the criteria are applied (identified in Table 1) and any associated waterbodies affected by water quality in those segments. The locations of the estuaries containing these segments are shown in Figure 1.

Figure 1. Relative Locations and Sizes of Estuary Segments Considered in this Opinion.

6 STATUS OF SPECIES LISTED AS ENDANGERED OR THREATENED UNDER THE ENDANGERED SPECIES ACT AND DESIGNATED CRITICAL HABITATS IN THE ACTION AREA AND UNDER THE JURISDICTION OF THE NATIONAL MARINE FISHERIES SERVICE

One cetacean, five sea turtle species, five species of fish, seven coral species, and one plant species inhabiting Florida waters that are listed as threatened or endangered are under NMFS' jurisdiction (Table 2). Designated critical habitat for four species occurs in Florida waters (North Atlantic right whale, loggerhead sea turtle, smalltooth sawfish, Atlantic sturgeon, and Johnson's seagrass).

The status for each species is discussed in the sections that follow with particular emphasis on aspects that may be influenced by Florida's water quality criteria for nutrients. Greater detail on species life history and status are available in the recovery plans and status reports for each species through the NMFS' Office of Protected Resources website: http://www.nmfs.noaa.gov/pr/species/esa/listed.htm.

This opinion applied the most recent recovery plans and status reports available at the time it was written. Note that recovery plans and status reports are periodically updated, so this content is not readily transferable to future assessments. While the following discussions focus on the use of Florida waters by these species, consideration was also given to the status of populations outside of the action area, which is important for evaluating how the risk to affected population(s) influences the status of the species as a whole.

Table 2. Endangered and Threatened Species and Designated Critical Habitat Under NMFS' Jurisdiction that Occur in Florida Waters.

Species	ESA Status	Designated critical habitat	Recovery Plan
Cetacean			
North Atlantic right whale (*Eubalaena glacialis*)	E – 35 FR 18319 & 73 FR 12024	63 FR 46693	70 FR 32293
Sea Turtles			
Green Turtle (*Chelonia mydas*)	E – 43 FR 32800	63 FR 46693	63 FR 28359
Hawksbill Turtle (*Eretmochelys imbricata)*	E – 35 FR 8491	63 FR 46693	57 FR 38818
Kemp's Ridley Turtle (*Lepidochelys kempii*)	E – 35 FR 18319	-- --	75 FR 12496
Leatherback Turtle (*Dermochelys coriacea*)	E – 61 FR 17	44 FR 17710	63 FR 28359
Loggerhead Turtle (*Caretta caretta*) – Northwest Atlantic DPS	E – 76 FR 58868	78 FR 39856	63 FR 28359
Fish			
Smalltooth sawfish (*Pristis pectinata*)	E – 68 FR 15674	74 FR 45353*	74 FR 3566
Shortnose sturgeon (*Acipenser brevirostrum*)	E – 32 FR4001	-- --	63 FR 69613
Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus*) South Atlantic DPS	E – 77 FR 5914	81 FR 36077 (proposed)	-- --
Gulf Sturgeon (*Acipenser oxyrinchus desotoi*)	T – 56 FR 49653	68 FR 13370	1995
Nassau grouper (*Epinephelus striatus*)	T – 81 FR 42268	-- --	biological report
Corals			
Elkhorn Coral (*Acropora palmata*)	T – 71 FR 26852	73 FR 72210	-- --
Staghorn Coral (*Acropora cervicornis*)	T – 71 FR 26852		-- --
Rough Cactus Coral *(Mycetophyllia ferox)*	T – 79 FR 54122	-- --	-- --
Pillar Coral (*Dendrogyra cylindrus)*	T – 79 FR 54122	-- --	-- --
Lobed Star Coral *(Orbicella annularis)*	T – 79 FR 54122	-- --	-- --
Mountainous Star Coral (*Orbicella faveolata)*	T – 79 FR 54122	-- --	-- --
Boulder Star Coral (*Orbicella franksi)*	T – 79 FR 54122	-- --	-- --
Marine Plant			
Johnson's Seagrass (*Halophila johnsonii*)	T – 63 FR 49035	65 FR 17786*	2002

*Designated critical habitat occurs in Florida

6.1 Cetacean: North Atlantic Right Whale

Description. The North Atlantic right whale is a stocky black bodied baleen whale. They weigh to up to 70 tons (140,000 lbs; 63,500 kg) with a length of about 50 feet (15 m). Calves are about 14 feet (4.2 m) at birth. The limited data available suggests that the life span of right whales is about 50 years.

Status. The North Atlantic right whale was originally listed as endangered in 1970. The western North Atlantic minimum stock size is based on a direct count of individual whales identified using photo-identification. The 25 October 2013 review of the photo-ID recapture database identified 465 individually recognized whales that were known to be alive during 2011. This number represents a minimum population size. The minimum population size calculated from the sightings database for the years 1990-2010 suggests a positive and slowly accelerating increase in population size.

Use of Florida Waters. North Atlantic right whale calving occurs from December through March in the coastal waters off Georgia and northern Florida. After calving, the adult females and calves migrate to northern feeding areas off the northeast U.S. and Canada. Most of the population, particularly the males and non-pregnant females, are not found in the calving area and may not follow this pattern (Morano et al. 2012). This species fasts during the winter and feeds during the summer, so the action is not expected to affect forage for this species.

Threats. Shark predation has been repeatedly documented on right whale calves along the southeastern U.S., some of which may be fatal (Taylor et al. 2013). Mortality or debilitation from infection or disease and red tide events are not known, but have the potential to be significant problems in the recovery of right whales because of their small population size. Historically, whaling was responsible for listing right whales as an endangered species. Currently, ship strikes and entanglement in commercial fishing gear pose the greatest threat to North Atlantic right whales. Infection of entanglement wounds can compromise health. Three quarters of 447 individuals examined between 1980 and 2002 showed scarring from fishing gear (Waring et al. 2013). Deaths of females are especially deleterious to the ability of the North Atlantic right whale population to recover. For instance, in 2005, ship strike mortalities included six adult females, three of which were carrying near-term fetuses and four of which were just starting to bear calves, thereby representing a lost reproductive potential of as many as 21 individuals over the short term (Kraus et al. 2005).

Climate-change associated shifts in calving intervals with sea surface temperature have already been documented for southern right whales (Leaper et al. 2006). The contribution of climate change to increased frequency of algal blooms was associated with the annual Southern right whale calf deaths. The calf deaths began in 2005 and strongly correlated with Harmful Algal Blooms (HABs). Calf death rates jumped from fewer than six per year prior to 2005 up to an average of 65 per year between 2005 and 2014. Exposure to algal toxins potentially occurred during gestation, by maternal transfer in milk or direct feeding. (Wilson et al. 2015).

North Atlantic right whales are exposed to toxic pollutants in their environment. Levels of chromium in North Atlantic right whale tissues are sufficient to be mutagenic and cause cell death in lung, skin, or testicular tissues (Wise et al. 2008, Chen et al. 2009). Flame retardants such as polybrominated diphenyl ethers (known to be carcinogenic) have also been detected in North Atlantic right whales (Montie et al. 2010). Mean PCB levels in North Atlantic right whales are greater than any other baleen whale species thus far measured (Van Scheppingen et al. 1996, Gauthier et al. 1997). Persistent pesticides and pesticide metabolites have been isolated from blubber samples (Woodley et al. 1991). The implication of these substances on the health and fitness of individuals is uncertain. Pesticides, although variable in concentration by season, do not appear to threaten North Atlantic right whale health and recovery (Weisbrod et al. 2000).

Designated critical habitat. In June of 1994, three habitat areas were designated as critical habitat for North Atlantic right whale feeding and calving. The designated critical habitats for feeding cover portions of the Great South Channel (east of Cape Cod), Massachusetts Bay and Cape Cod Bay, and Stellwagen Bank. The designated critical habitat area protecting calving and breeding grounds is along Georgia and northeastern Florida coasts (Figure 2). These whales calve and breed in shallow coastal waters. This designated critical habitat has generally fared better than northern designated critical habitat and significant degradation has not been clearly identified (Keller et al. 2012).

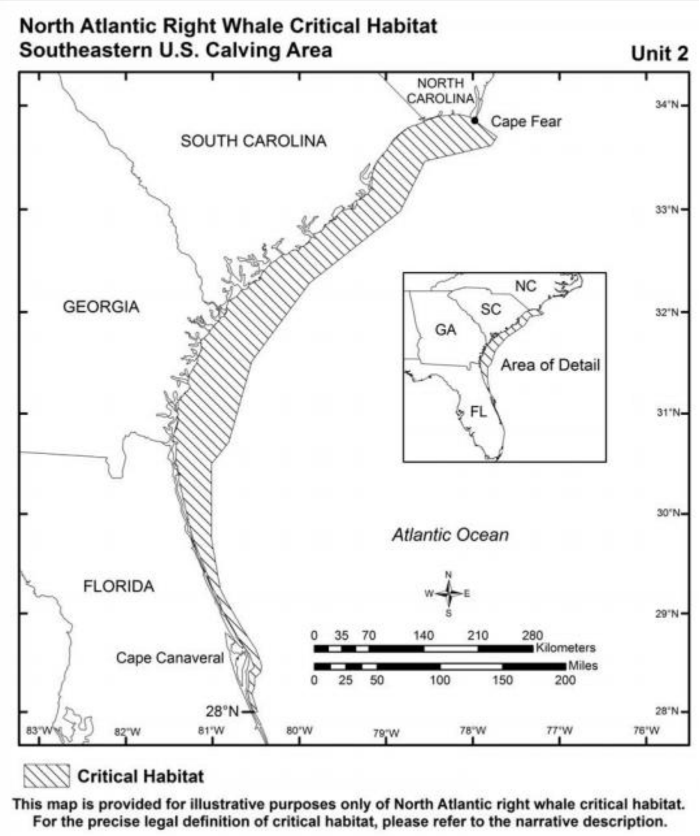

Figure 2. North Atlantic Right Whale Designated Critical Habitat Along the Southeastern Coast.

6.2 Sea Turtles

A number of threats are common to all sea turtles.[2] Predation is a primary natural threat. While cold stunning is not a major concern for leatherback sea turtles, which can tolerate low water temperatures, it is considered a major natural threat to other sea turtle species. Disease is also a factor in sea turtle survival. Fibropapillomatosis (FP) tumors are a major threat to green turtles in some areas of the world, and tumor occurrence is particularly associated with degraded coastal habitat. Scientists have also documented FP in populations of loggerhead, olive ridley, and flatback turtles, but reports in green turtles are more common. Large tumors can interfere with feeding and essential behaviors, and tumors on the eyes can cause permanent blindness. FP was first described in green turtles in the Florida Keys in the 1930s. Since then it has been recorded in many green turtle populations around the world. The effects of FP at the population level are not well understood. The sand-borne fungal pathogens *Fusarium falciforme* and *F. keratoplasticum* are capable of killing greater than 90 percent of sea turtle embryos they infect, threatening nesting productivity under some conditions. These pathogens can survive on decaying organic matter and embryo mortality rates attributed to fusarium were associated with clay/silt nesting areas compared to sandy areas (Sarmiento-Ramırez et al. 2014). Brevitoxin has recently been identified as a potential FP tumor promoter (Perrault et al. 2017).

Sea turtles breathe air and are only exposed to pollutants through drinking water and very limited absorption through mucous membranes. As discussed previously, excess nutrients, the stressors of concern for this opinion (Section 8.1.1), can lead to elevated ammonia. Ingested ammonia is of low toxicity for many species (World Health Organization 1996, Agency for Toxic Substances and Disease Registry 2004). Toxicity data for sea turtles and other reptiles, both freshwater and marine, are limited. One study found that examined ammonia effects on the freshwater soft-shelled turtle, *Pelodiscus sinensis*: Exposure of juveniles to very high ammonia concentrations of up to 90 milligrams of total ammonia nitrogen per liter in ambient water for 97 days (i.e., exposure was both through drinking and dermal absorption) had no influence on energy budget or metabolism (Jing and Niu 2008).

Fishing is the primary anthropogenic threat to sea turtles in the ocean. Fishing gear entanglement potentially drowns or seriously injures sea turtles. Fishing dredges can crush and entrap turtles, causing death and serious injury. Infection of entanglement wounds can compromise health. The development and operation of marinas and docks in inshore waters can negatively affect nearshore habitats. Turtles swimming or feeding at or just beneath the surface of the water are particularly vulnerable to boat and vessel strikes, which can result in serious propeller injuries and death.

Ingestion or entanglement in marine debris is a cause of morbidity and mortality for sea turtles in the pelagic (open ocean) environment (Stamper et al. 2009). Consumption of non-nutritive debris also reduces the amount of nutritive food ingested, which then may decrease somatic growth and

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[2] See http://www.nmfs.noaa.gov/pr/species/turtles/threats.htm, updated June 16, 2014

reproduction (McCauley and Bjorndal 1999). Marine debris is especially problematic for turtles that spend all or significant portions of their life cycle in the pelagic environment (e.g., leatherbacks, juvenile loggerheads, and juvenile green turtles).

Sea turtle nesting and marine environments are facing increasing impacts through structural modifications, sand nourishment, and sand extraction to support widespread development and tourism (Lutcavage et al. 1997, Bouchard et al. 1998, Hamann et al. 2006, Maison 2006, Hernandez et al. 2007, Santidrián Tomillo et al. 2007, Patino-Martinez 2013). These factors decrease the amount of nesting area available to nesting females, and may evoke a change in the natural behaviors of adults and hatchlings through direct loss of and indirect (e.g., altered temperatures, erosion) mechanisms (Ackerman 1997, Witherington et al. 2003, Hirama et al. 2007). Lights from developments alter nesting adult behavior and are often fatal to emerging hatchlings as they are drawn to light sources and away from the sea (Witherington and Bjorndal 1991, Witherington 1992, Cowan et al. 2002, Deem et al. 2007, Bourgeois et al. 2009).

Beach nourishment also affects the incubation environment and nest success. Although the placement of sand on beaches may provide a greater quantity of nesting habitat, the quality of that habitat may be less suitable than pre-existing natural beaches. Constructed beaches tend to differ from natural beaches in several important ways. They are typically wider, flatter, more compact, and the sediments are more moist than those on natural beaches (Nelson et al. 1987, Ackerman 1997, Ernest and Martin 1999). Nesting success typically declines for the first year or two following construction, even when more nesting area is available for turtles (Trindell et al. 1998, Ernest and Martin 1999, Herren 1999). Likely causes of reduced nesting success on constructed beaches include increased sand compaction, escarpment formation, and changes in beach profile (Nelson et al. 1987, Grain et al. 1995, Lutcavage et al. 1997, Steinitz et al. 1998, Ernest and Martin 1999, Rumbold et al. 2001). Compaction can inhibit nest construction or increase the amount of time it takes for turtles to construct nests, while escarpments often cause female turtles to return to the ocean without nesting or to deposit their nests seaward of the escarpment where they are more susceptible to frequent and prolonged tidal inundation. In short, sub-optimal nesting habitat may cause decreased nesting success, place an increased energy burden on nesting females, result in abnormal nest construction (Carthy 2006), and reduce the survivorship of eggs and hatchlings. In addition, sand used to nourish beaches may have a different composition than the original beach; thus introducing lighter or darker sand, consequently affecting the relative nest temperatures (Ackerman 1997, Milton et al. 1997).

In addition to effects on sea turtle nesting habitat, anthropogenic disturbances also threaten coastal foraging habitats, particularly areas rich in seagrass and marine algae. Coastal habitats are degraded by pollutants from coastal runoff, marina and dock construction, dredging, aquaculture, oil and gas exploration and extraction, increased under water noise and boat traffic, as well as structural degradation from excessive boat anchoring and dredging (Francour et al. 1999, Lee Long et al. 2000, Waycott et al. 2005).

Conant et al. (2009) included a review of the impacts of marine pollutants on sea turtles: marine debris, oil spills, and bioaccumulative chemicals. Sea turtles at all life stages appear to be highly sensitive to oil spills, perhaps due to certain aspects of their biology and behavior, including a lack of avoidance behavior, indiscriminate feeding in convergence zones, and large pre-dive inhalations (Milton et al. 2003). Milton et al. (2003) state that the oil effects on turtles include increased egg mortality and developmental defects, direct mortality due to oiling in hatchlings, juveniles and adults, and impacts to the skin, blood, salt glands, and digestive and immune systems. Vargo et al. (1986) reported that sea turtles would be at substantial risk if they encountered an oil spill or large amounts of tar in the environment. In a review of available information on debris ingestion, Balazs (1985) reported that tar balls were the second most prevalent type of debris ingested by sea turtles. Physiological experiments showed that sea turtles exposed to petroleum products may suffer inflammatory dermatitis, ventilator disturbance, salt gland dysfunction or failure, red blood cell disturbances, immune response, and digestive disorders (Vargo et al. 1986, Lutz and Lutcavage 1989, Lutcavage et al. 1995).

Conant et al. (2009) describes the potentially extensive impacts of climate change on all aspects of a sea turtle's life cycle, as well as impact the abundance and distribution of prey items. Rising sea level is one of the most certain consequences of climate change (Titus and Narayanan 1995), and will result in increased erosion rates along nesting beaches. This could particularly affect areas with low-lying beaches where sand depth is a limiting factor, as the sea will inundate nesting sites and decrease available nesting habitat (Daniels et al. 1993, Fish et al. 2005, Baker et al. 2006). The loss of habitat because of climate change could be accelerated due to a combination of other environmental and oceanographic changes such as an increase in the frequency of storms and/or changes in prevailing currents, both of which could lead to increased beach loss via erosion (Baker et al. 2006). On some undeveloped beaches, shoreline migration will have limited effects on the suitability of nesting habitat. The Bruun rule specifies that during a sea level rise, a typical beach profile will maintain its configuration but will be translated landward and upward (Rosati et al. 2013). However, along developed coastlines, and especially in areas where erosion control structures have been constructed to limit shoreline movement, rising sea levels will cause severe effects on nesting females and their eggs. Erosion control structures can result in the permanent loss of dry nesting beach or deter nesting females from reaching suitable nesting sites (NRC 1990a). Nesting females may deposit eggs seaward of the erosion control structures potentially subjecting them to repeated tidal inundation. Non-native vegetation often out competes native species, is usually less stabilizing, and can lead to increased erosion and degradation of suitable nesting habitat. Exotic vegetation may also form impenetrable root mats that can prevent proper nest cavity excavation, invade and desiccate eggs, or trap hatchlings.

Threats in the Southeast United States. In the southeastern U.S., numerous erosion control structures that create barriers to nesting have been constructed. The proportion of coastline that is armored is approximately 18 percent (239 km) in Florida (Schroeder and Mosier 1998, Witherington et al. 2006), In the Northwest Atlantic, jetties have been placed at many ocean inlets to keep transported sand from closing the inlet channel. Erosion of Northwest Atlantic beaches and dunes is accelerated by sheet flow, through stormwater outfalls, or through small diameter pipes. These outfalls create localized erosion channels, prevent natural dune establishment, and wash out sea turtle nests (FDEP 2008, Humiston & Moore Engineers 2010). Contaminants contained in stormwater, such as oils, grease, antifreeze, gasoline, metals, pesticides, chlorine, and nutrients, are discharged onto the beach. Reports of hatchling disorientation events[3] in Florida alone describe several hundred nests each year and are likely to involve tens of thousands of hatchlings (Nelson et al. 2002). However, this number calculated from disorientation reports is likely a vast underestimate. Independent of these reports, Witherington et al. (1995) surveyed hatchling orientation at nests located at 23 representative beaches in six counties around Florida in 1993 and 1994 and found that, by county, approximately 10 to 30 percent of nests showed evidence of hatchlings disoriented by lighting.

6.2.1 Green Sea Turtle

Description. Green sea turtles have a smooth shell with shades of black, gray, green, brown, and yellow; their bottom shell is yellowish white. Adults weigh 300-350 pounds (135-150 kg) and measure 3 feet in length. Hatchlings weigh 0.05 pounds (25 g) and are 2 inches (50 mm) long. Growth rates of juveniles vary substantially among populations, ranging from <1 cm/year (Green 1993) to >5 cm/year (McDonald Dutton and Dutton 1998), likely due to differences in diet quality, duration of foraging season (Chaloupka et al. 2004), and density of turtles in foraging areas (Bjorndal and Bolten 2000, Seminoff et al. 2002b, Balazs and Chaloupka 2004).

Status. Federal listing of the green sea turtle was published July 28, 1978. The Florida and Pacific coast of Mexico breeding populations were listed as endangered and all other populations were listed as threatened. Overall, of the 26 sites for which data enable an assessment of current trends, 12 nesting populations are increasing, 10 are stable, and four are decreasing. The most important nesting concentration for green sea turtles in the western Atlantic is in Tortuguero, Costa Rica where nesting has increased considerably since the 1970s (NMFS and USFWS 2007a). Trend data should be interpreted cautiously because data are only available for just over half of all sites examined and very few data sets span a full green sea turtle generation (Seminoff 2004). Over the course of a long-term study along Cape Canaveral, Florida, average turtle length and recapture rate both declined (Redfoot and Ehrhart 2013).

Use of Florida Waters. The vast majority of green sea turtle nesting within the southeastern U.S. occurs in Florida (Johnson and Ehrhart 1994, Meylan et al. 1995). Nesting has been increasing since 1989 (Florida Fish and Wildlife Conservation Commission-FWCC, Florida Marine Research Institute Index Nesting Beach Survey Database) with biennial peaks in abundance and a generally positive trend during the ten years of regular monitoring. This includes the Atlantic coast of Florida on beaches where only loggerhead nesting was observed in the past (Pritchard 1997). Recent modeling by (Chaloupka et al. 2008a) using data sets of 25

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[3] Hatchlings orienting away from the ocean and towards artificial light.

years or more has resulted in an estimate of the Florida nesting stock at the Archie Carr National Wildlife Refuge growing at an annual rate of 13.9 percent, and the Tortuguero, Costa Rica, population growing at 4.9 percent.

There are no reliable estimates of the number of immature green sea turtles that inhabit coastal areas of the southeastern U.S. However, the annual number of incidental captures of immature green sea turtles by offshore cooling water intake structures at the St. Lucie Power Plant in Florida shows a significant increase. Captures averaged 19 for 1977-1986, 178 for 1987-1996, and 262 for 1997-2001 (Florida Power and Light Company St. Lucie Plant 2002). More recent unpublished data shows 101 captures in 2007, 299 in 2008, 38 in 2009 (power output was cut—and cooling water intake concomitantly reduced—for part of that year) and 413 in 2010. Ehrhart et al. (2007) documented a significant increase in the in-water abundance of green turtles in the Indian River Lagoon area.

Habitat and Forage. Once hatched, turtles enter the sea to live a pelagic phase preferentially in drift lines or surface current convergences, probably because of the prevalence of cover and higher prey densities that associate with flotsam (NMFS and USFWS 1998a). At approximately 20-25 cm carapace length, juveniles leave pelagic habitats, enter benthic foraging areas (Bjorndal 1997), and spend the majority of their lives in coastal foraging grounds (MacDonald et al. 2012). These areas include both open coastline and protected bays and lagoons. While offshore and sometimes in coastal habitats, green sea turtles are not obligate plant-eaters as widely believed, and instead consume invertebrates such as jellyfish, sponges, sea pens, and pelagic prey (Godley et al. 1998, Heithaus et al. 2002, Seminoff et al. 2002a, Parker and Balazs 2005, Hatase et al. 2006, Hart et al. 2013). A shift to a more herbivorous diet occurs when individuals move into neritic habitats (i.e., sandy, muddy bottoms) (Cardona et al. 2010).

The largely plant-eating diet of green turtles is believed to be responsible for their particularly slow growth rates (Bjorndal 1982). If individuals do not feed sufficiently, growth is stunted and apparently does not compensate even when greater-than-needed resources are available (Roark et al. 2009). There is some evidence that individuals move from shallow seagrass beds during the day to deeper areas at night (Hazel 2009).

Threats. Adult survivorship is lower in areas of human impact on green sea turtles and their habitats (Bjorndal et al. 2003, Campbell and Lagueux 2005). Green sea turtles with an abundance of barnacles have been found to have a much greater probability of having health issues (Flint et al. 2009). Major anthropogenic impacts to the nesting and marine environment affect green sea turtle survival and recovery (Patino-Martinez 2013). Hundreds of mostly immature green sea turtles were killed between 2006 and 2008 due to bycatch and direct harvest along Baja California Sur (Senko et al. 2014). Green sea turtles stranded in Brazil were all found to have ingested plastics or fishing debris (n=34). Ingested debris appeared to be the direct cause of mortality in three of the 34 animals (Tourinho et al. 2009). The introduction of alien algae species threatens the stability of some coastal ecosystems and may lead to the elimination of preferred dietary species of green sea turtles (De Weede 1996). Very few green sea turtles are bycaught in U.S. fisheries (Finkbeiner et al. 2011). Fuentes et al. (2010) predicted that rising temperatures due to climate change would be a much greater threat in the long term to the hatching success of sea turtles in general and green sea turtles along northeastern Australia particularly. Green sea turtles emerging from nests at cooler temperatures likely absorb more yolk that is converted to body tissue than do hatchlings from warmer nests (Ischer et al. 2009). Predicted temperature rises may approach or exceed the upper thermal tolerance limit of sea turtle incubation, causing widespread failure of nests (Fuentes et al. 2010).

Chlordane, lindane, endrin, endosulfan, dieldrin, DDT and PCB have been detected in green sea turtle tissues (Miao et al. 2001, Gardner et al. 2003). The DDT metabolite, DDE, has not been found to influence sex determination at levels below cytotoxicity (Podreka et al. 1998, Keller and McClellan-Green 2004). Flame retardants have been measured in tissues from healthy individuals (Hermanussen et al. 2008). Copper, lead, manganese, cadmium, and nickel have been found in various tissues and life stages (Barbieri 2009). Arsenic also occurs in very high levels in green sea turtle eggs (van de Merwe et al. 2009). Exposure to sewage effluent may result in green sea turtle eggs harboring antibiotic-resistant strains of bacteria (Al-Bahry et al. 2009). To date, no tie has been found between pesticide concentration and susceptibility to FP, although degraded habitat and pollution have been tied to the incidence of the disease in green turtle (Aguirre et al. 1994, Foley et al. 2005). It has also been theorized that exposure to macroalgae from eutrophic environments (Van Houtan et al. 2014), tumor-promoting compounds produced by cyanobacteria *Lyngbya majuscule* (Arthur et al. 2008) or dinoflagellates of the genus *Prorocentrum* (Landsberg et al. 1999) promote the development of FP.

Designated critical habitat. On September 2, 1998, designated critical habitat for green sea turtles was designated in coastal waters surrounding Culebra Island, Puerto Rico. Aspects of these areas that are important for green sea turtle survival and recovery include important natal development habitat, refuge from predation, shelter between foraging periods, and seagrasses, which are the principal dietary component of juvenile and adult green turtles throughout the Wider Caribbean region (Bjorndal 1997).

6.2.2 Hawksbill Sea Turtle

Description. Hawksbill sea turtles have a dark to golden brown shell, with streaks of orange, red, and/or black with a serrated back and overlapping "scutes," while the bottom shell (plastron) is clear yellow. Hatchlings are mostly brown. Adults weigh up to 100-150 pounds (45-70 kg) and measure 25-35 inches (65-90 cm) long. Hatchlings weigh 0.5 ounces (15 g). Within United States territories and U.S. dependencies in the Caribbean Region, hawksbill sea turtles nest principally in Puerto Rico and the U.S. Virgin Islands, particularly on Mona Island and Buck Island.

Status. Hawksbill sea turtles received protection on June 2, 1970 under the Endangered Species Conservation Act and since 1973 have been listed as endangered under the ESA. Although no historical records of abundance are known, hawksbill sea turtles are considered to be severely depleted due to the fragmentation and low use of current nesting beaches (NMFS and USFWS

2007b). Among 42 sites for which recent trend data are available, 10 (24 percent) are increasing, three (7 percent) are stable and 29 (69 percent) are decreasing.

Use of Florida Waters. Hawksbill sea turtles appear to be rare visitors to the U.S. Gulf of Mexico, with Florida being the only Gulf state with regular sightings (Rabalais and Rabalais 1980, Hildebrand 1983, Witzell 1983, NMFS and USFWS 1993, Rester and Condrey 1996). Within the continental United States, hawksbill sea turtles nest only on beaches along the southeast coast of Florida and in the Florida Keys.

Habitat and Forage. Hawksbill sea turtles use a wide range habitats during their lifetimes (Musick and Limpus 1997, Plotkin 2003). After hatching, hawksbills are pelagic, associated with sargassum (Musick and Limpus 1997) until they are approximately 22-25 cm in straight carapace length (Meylan 1988, Meylan and Donnelly 1999). In the post pelagic phase, they inhabit coral reefs or other hard-bottom habitats, seagrass, algal beds, mangrove bays and creeks (Musick and Limpus 1997, Bjorndal and Bolten 2010), and mud flats (R. von Brandis, unpublished data in NMFS and USFWS 2007b (NMFS and USFWS 2007b)). Dietary data from oceanic stage hawksbills are limited, but indicate a combination of plant and animal material (Bjorndal 1997). Sponges and octocorals are common prey off Honduras (Berube et al. 2012, Hart et al. 2013).

Threats. One natural threat unique to hawksbill sea turtles is hybridization (Mortimer and Donnelly 2008) with other species of sea turtles. Future impacts from climate change and global warming may result in significant changes in hatchling sex ratios. The fact that hawksbill turtles exhibit temperature-dependent sex determination (Wibbels 2003) suggests that there may be a skewing of future hawksbill cohorts toward strong female bias (since warmer temperatures produce more female embryos).

Designated critical habitat. On September 2, 1998, NMFS designated critical habitat for hawksbill sea turtles around Mona and Monito Islands, Puerto Rico. Aspects of these areas that are important for hawksbill sea turtle survival and recovery include important natal development habitat, refuge from predation, shelter between foraging periods, and food for hawksbill sea turtle prey.

6.2.3 Kemp’s Ridley Sea Turtle

Description. The Kemp's ridley sea turtle has a grayish-green, nearly circular, top shell with a pale yellowish bottom shell. Adults weight 100 pounds (45 kg) and measure 24-28 inches (60-70 cm) in length. Each of the front flippers has one claw while the back flippers may have one or two. Hatchlings weigh 0.5 ounces (14 g) and are 1.5 inches (3.8 cm) long.

Status. The Kemp’s ridley sea turtle was listed as endangered on December 2, 1970. Internationally, the Kemp’s ridley is considered the most endangered sea turtle (NRC 1990b, USFWS 1999). Historic information indicates that tens of thousands of Kemp’s ridleys nested near Rancho Nuevo, Mexico, during the late 1940s (Hildebrand 1963). From 1978 through the 1980s, arribadas involved 200 turtles or less, and by 1985, the total number of nests at Rancho Nuevo had dropped to approximately 740 for the entire nesting season, or a projection of roughly

234 turtles (USFWS and NMFS 1992, TEWG 2000). Beginning in the 1990s, an increasing number of beaches in Mexico were being monitored for nesting, and the total number of nests on all beaches in Tamaulipas and Veracruz in 2002 was over 6,000; the rate of increase from 1985 ranged from 14-16 percent (TEWG 2000, USFWS 2002, Heppell et al. 2005). Preliminary estimates of 2011 and 2012 nesting support 19,368 and 20,197 nests, respectively (Gallaway et al. 2013). Gallaway et al. (2013) estimated that nearly 189,000 female Kemp’s ridley sea turtles over the age of two years were alive in 2012. Extrapolating based upon sex bias, the authors estimated that nearly a quarter million age two, or older Kemp’s ridleys were alive at this time.

Use of Florida Waters. The vast majority of individuals stem from breeding beaches at Rancho Nuevo on the Gulf of Mexico coast of Mexico. The migratory corridors appear to extend throughout the coastal areas of the Gulf of Mexico and most turtles appear to travel in waters less than roughly 164 feet in-depth. Turtles that headed north and east traveled as far as southwest Florida, whereas those that headed south and east traveled as far as the Yucatan Peninsula, Mexico (Morreale et al. 2007). Kemp’s ridleys in south Florida begin to migrate northward during spring toward Long Island Sound and even Nova Scotia in late summer (Bleakney 1955), returning south in the winter as local water temperatures cool (Lutcavage and Musick 1985, Byles 1988, Keinath 1993, Renaud 1995). They reside in winter-feeding areas for several months (Byles and Plotkin 1994, Morreale et al. 2007). During spring and summer, juvenile Kemp’s ridleys occur in the shallow coastal waters of the northern Gulf of Mexico from south Texas to north Florida. In the fall, most Kemp’s ridleys migrate to deeper or more southern warmer waters and remain there through the winter (Schmid 1998). As adults, many turtles remain in the Gulf of Mexico, with only occasional occurrence in the Atlantic Ocean (NMFS et al. 2010).

Habitat and Forage. Developmental habitats for juveniles occur throughout the entire coastal Gulf of Mexico and U.S. Atlantic coast northward to New England (Schmid 1998, Wibbels et al. 2005, Morreale et al. 2007). Key foraging areas in the Gulf of Mexico include Sabine Pass, Texas; Caillou Bay and Calcasieu Pass, Louisiana; Big Gulley, Alabama; Cedar Keys, Florida; and Ten Thousand Islands, Florida (Carr and Caldwell 1956, Ogren 1989, Coyne et al. 1995, Schmid 1998, Schmid et al. 2002, Witzell and Schmid 2005). Foraging areas studied along the Atlantic coast include Pamlico Sound, Chesapeake Bay, Long Island Sound, Charleston Harbor, and Delaware Bay. Near-shore waters of 120 feet or less provide the primary marine habitat for adults, although it is not uncommon for adults to venture into deeper waters (Byles 1989, Mysing and Vanselous 1989, Renaud et al. 1996, Shaver et al. 2005, Shaver and Wibbels 2007). Benthic coastal waters of Louisiana and Texas seem to be preferred foraging areas for Kemp’s ridley sea turtles (particularly passes and beachfronts), although individuals may travel along the entire coastal margin of the Gulf of Mexico (Renaud 1995, Landry et al. 1996, Landry and Costa 1999). Kemp’s ridley diet consists mainly of swimming crabs, but may also include fish, jellyfish, and an array of mollusks. Immature Kemp’s ridleys off southwest Florida documented predation on benthic tunicates, a previously undocumented food source for this species (Witzell and Schmid 2005).

Threats. Kemp's ridley sea turtles are particularly prone to cold stunning along Cape Cod (Innis et al. 2009). Habitat destruction remains a concern in the form of bottom trawling and shoreline development. Trawling destroys habitat utilized by Kemp's ridley sea turtles for feeding and construction activities can produce hazardous runoff. The vast majority of fisheries interactions with sea turtles in the U.S. are either Kemp's ridleys or loggerhead sea turtles (Finkbeiner et al. 2011). Roughly three-quarters of annual mortality was attributed to shrimp trawling prior to Turtle Exclusion Device regulations (Gallaway et al. 2013). However, this has dropped to an estimated one-quarter of total mortality nearly 20 years after Turtle Exclusion Device Turtle Exclusion Devices were implemented in 1990 (Gallaway et al. 2013).

Designated critical habitat. NMFS has not designated critical habitat for Kemp's ridley sea turtle.

6.2.4 Leatherback Sea Turtle

Description. Leatherback sea turtles have a primarily black shell with pinkish-white coloring on their belly. A leatherback's top shell (carapace) is about 1.5 inches (4 cm) thick and consists of leathery, oil-saturated connective tissue overlaying loosely interlocking dermal bones. Their carapace has seven longitudinal ridges and tapers to a blunt point, which help give the carapace a more hydrodynamic structure. Adults weigh up to 2,000 pounds (900 kg) and measure 6.5 feet (2 m) long. Hatchlings weigh 1.5-2 ounces (40-50 g) and are 2-3 inches (50-75 cm) in length.

Status. Leatherback sea turtles initially received protection on June 2, 1970 under the Endangered Species Conservation Act and, since 1973, have been listed as endangered under the ESA. In the Caribbean, Atlantic, and Gulf of Mexico, leatherback populations are generally increasing. In the U.S., nesting in Puerto Rico, St. Croix, and the U.S. Virgin Islands continues to increase as well, with some shift in the nesting between these two islands.

Use of Florida Waters. Florida's Atlantic coast is one of the main nesting areas in the continental U.S. Data from this area reveals a fluctuating, but general upward trend. Florida index nesting beach data from 1989-2014, indicate that number of nests at core index nesting beaches ranged from 27 to 641 in 2014. Leatherback sea turtles feed in shallow waters on the continental shelf waters along the Florida Panhandle, the Mississippi River Delta, and the Texas coast (Collard 1990). Leatherbacks occur along the southeastern U.S. year-round, with peak abundance in summer (TEWG 2007). In spring, leatherback sea turtles appear to be concentrated near the coast, while other times of the year they are spread out at least to the Gulf Stream.

Habitat and Forage. Leatherbacks are primarily pelagic but occur throughout marine waters including nearshore habitats (Schroeder and Thompson 1987, Shoop and Kenney 1992, Grant and Ferrell 1993, Starbird et al. 1993). After nesting, female leatherbacks migrate from tropical waters to more temperate latitudes, which support high densities of jellyfish prey in the summer.

Threats. Plastic ingestion is very common in leatherbacks, blocking gastrointestinal tracts and potentially leading to death (Mrosovsky et al. 2009). Egg collection is widespread and attributed to catastrophic declines, such as in Malaysia. Harvest of females along nesting beaches is of concern worldwide. Bycatch, particularly by longline fisheries, is a major source of mortality for leatherback sea turtles (Crognale et al. 2008, Gless et al. 2008, Fossette et al. 2009, Petersen et al. 2009).

Designated critical habitat. On March 23, 1979, leatherback designated critical habitat was identified adjacent to Sandy Point, St. Croix, U.S. Virgin Islands from the 183 m isobath to mean high tide level between 17° 42’12” N and 65°50’00” W. This habitat is essential for nesting, which has been increasingly threatened since 1979, when tourism increased significantly, bringing nesting habitat and people into close and frequent proximity. However, studies do not currently support significant designated critical habitat deterioration. On January 26, 2012, NMFS designated critical habitat for leatherback sea turtles in waters along Washington State and Oregon (Cape Flattery to Cape Blanco; 64,760 square kilometers) and California (Point Arena to Point Arguello; 43,798 square kilometers).

6.2.5 Northwest Atlantic Distinct Population Segment of the Loggerhead Sea Turtle

Description. Adult loggerhead sea turtles have relatively large heads, which support powerful jaws. They have a reddish-brown, slightly heart-shaped top shell with pale yellowish bottom shell. The neck and flippers are usually dull brown to reddish brown on top and medium to pale yellow on the sides and bottom. They weigh 250 pounds (113 kg) and measure 3 feet (~1 m) in length. Hatchlings are brown to dark gray with a yellowish to tan bottom shell. Their flippers are dark gray to brown above with white-to-white-gray margins. They weigh 0.05 pounds (20 g) and are 2 inches (4 cm) long.

Status. Loggerhead sea turtles were originally listed as threatened under the ESA on July 28, 1978. On September 22, 2011, NMFS designated nine DPSs of loggerhead sea turtles as threatened. The global abundance of nesting female loggerhead turtles was estimated at 43,320–44,560 (Spotila 2004).

Use of Florida Waters. The greatest concentration of loggerheads occurs in the Atlantic Ocean and the adjacent Caribbean Sea, primarily on the Atlantic coast of Florida, with other major nesting areas located on the Yucatán Peninsula of Mexico, Columbia, Cuba, South Africa (Márquez 1990, LGL Ltd. 2007).

Because of its size, the south Florida subpopulation of loggerheads may be critical to the survival of the species in the Atlantic, and in the past it was considered second in size only to the Oman nesting aggregation (NMFS and USFWS 1991). The South Florida population increased at ~5.3 percent per year from 1978-1990, and was initially increasing at 3.9-4.2 percent after 1990. An analysis of nesting data from 1989-2005, a period of more consistent and accurate surveys than in previous years, showed a detectable trend and, more recently (1998-2005), has shown evidence of a declining trend of approximately 22.3 percent (FFWCC 2007b, a, Witherington et al. 2009). This is likely due to a decline in the number of nesting females within the population (Witherington et al. 2009). Nesting data from the Archie Carr Refuge (one of the most important nesting locations in southeast Florida) over the last 6 years shows nests declined from approximately 17,629 in 1998 to 7,599 in 2004, also suggesting a decrease in population size. While this is a long period of decline relative to the past-observed nesting pattern at this location, aberrant ocean surface temperatures complicate the analysis and interpretation of these data. Although caution is warranted in interpreting the decreasing nesting trend given inherent annual fluctuations in nesting and the short time period over which the decline has been noted, the recent nesting decline at this nesting beach is reason for concern. Loggerhead nesting is thought to consist of just 60 nesting females in the U.S. Caribbean and U.S.Gulf of Mexico. Data from several studies showed decreased growth rates of loggerheads in U.S. Atlantic waters from 1997-2007, corresponding to a period of 43 percent decline in Florida nest counts (Bjorndal et al. 2013).

Loggerheads associated with the South Florida nesting aggregation occur in higher frequencies in the Gulf of Mexico (where they represent ~10 percent of the loggerhead captures) and the Mediterranean Sea (where they represent ~45 percent of loggerhead sea turtles captured). In the North Atlantic, loggerheads travel north during spring and summer as water temperatures warm and return south in fall and winter, but occur offshore year-round assuming adequate temperature. Satellite tracking of loggerheads from southeastern U.S. nesting beaches supports three dispersal modes to foraging areas: one northward along the continental shelf to the northeastern U.S., broad movement through the southeastern and mid-Atlantic U.S., and residency in areas near breeding areas (Reina et al. 2012).

An estimated 12 percent of all western North Atlantic Ocean loggerhead sea turtles reside in the eastern Gulf of Mexico, with the vast majority in western Florida waters (Davis et al. 2000, TEWG 2009). Loggerheads may occur in both offshore habitats (particularly around oil platforms and reefs, where prey and shelter are available; (Fritts et al. 1983, Rosman et al. 1987, Lohoefener et al. 1990, Gitschlag and Herczeg 1994, Davis et al. 2000), as well as shallow bays and sounds (which may be important developmental habitat for late juveniles in the eastern Gulf of Mexico; (Lohoefener et al. 1990, USAF 1996, Davis et al. 2000).

Habitat and Forage. Loggerhead sea turtles are omnivorous and opportunistic feeders through their lifetimes (Parker and Balazs 2005). Hatchling loggerheads migrate to the ocean, where they are generally believed to lead a pelagic existence for as long as 7-12 years (Avens et al. 2013) feeding on macroplankton associated with *Sargassum* spp. communities (NMFS and USFWS 1991). Pelagic and benthic juveniles forage on crabs, mollusks, jellyfish, and vegetation at or near the surface (Dodd 1988, Wallace et al. 2009). Sub-adult and adult loggerheads prey on benthic invertebrates such as gastropods, mollusks, and decapod crustaceans in hard-bottom habitats, although fish and plants are also occasionally eaten (NMFS and USFWS 1998b). Stable isotope analysis and study of organisms on turtle shells has recently shown that although a loggerhead population may feed on a variety of prey, individuals composing the population have specialized diets (Reich et al. 2010, Vander Zanden et al. 2010).

Threats. High temperatures before hatchlings emerge from their nests can reduce hatchling success, as can bacterial contamination and woody debris in nests (Trocini 2013). Brevetoxin-producing algal blooms can result in loggerhead sea turtle death and pathology, with nearly all stranded loggerheads in affected areas showing signs of illness or death resulting from exposure (Fauquier et al. 2013). Shrimp trawl fisheries account for the highest number of captured and killed loggerhead sea turtles. Along the Atlantic coast of the U.S., NMFS estimated that shrimp trawls capture almost 163,000 loggerhead sea turtles each year in the Gulf of Mexico, of which 3,948 die. However, more recent estimates from suggest interactions and mortality has decreased from pre-regulatory periods, with a conservative estimate of 26,500 loggerheads captured annually in U.S. Atlantic fisheries causing mortality up to 1,400 individuals per year (Finkbeiner et al. 2011). Commercial gillnet fisheries are estimated to have killed 52 loggerheads annually along the U.S. mid-Atlantic (Murray 2013). Pacific bycatch is much less, with about 400 individuals bycaught annually in U.S. fisheries resulting in at least 20 mortalities (Finkbeiner et al. 2011). Offshore longline tuna and swordfish longline fisheries are also a serious concern for the survival and recovery of loggerhead sea turtles and appear to affect the largest individuals more than younger age classes (Bolten et al. 1994, Aguilar et al. 1995, Howell et al. 2008, Tomás et al. 2008, Carruthers et al. 2009, Marshall et al. 2009, Petersen et al. 2009). Longline hooking along Hawaii and California suggests a 28 percent mortality rate for hooked and released loggerheads, with no significant difference between shallow- versus deep-hooked individuals (Swimmer et al. 2013). Deliberate hunting of loggerheads for their meat, shells, and eggs has declined from previous exploitation levels, but still exists and hampers recovery efforts (Lino et al. 2010).

More than one-third of loggerheads found stranded or bycaught had ingested marine debris in a Mediterranean study, with possible mortality resulting in some cases (Lazar and Gračan 2010). Another study in the Tyrrhenian Sea found 71 percent of stranded and bycaught sea turtles had plastic debris in their guts (Campani et al. 2013). Another threat marine debris poses is to hatchlings on beaches escaping to the sea. Two thirds of loggerheads contacted marine debris on their way to the ocean and many became severely entangled or entrapped by it (Triessnig et al. 2012).

Climate change may also have significant implications on loggerhead populations worldwide. Loggerhead sea turtles are very sensitive to temperature as a determinant of sex while incubating. Ambient temperature increase by just 1º-2º C can potentially change hatchling sex ratios to all or nearly all female in tropical and subtropical areas (Hawkes et al. 2007). Over time, this can reduce genetic diversity, or even population viability, if males become a small proportion of populations (Hulin et al. 2009). Sea surface temperatures on loggerhead foraging grounds correlate to the timing of nesting, with higher temperatures leading to earlier nesting (Mazaris et al. 2009, Schofield et al. 2009). Increasing ocean temperatures may also lead to reduced primary productivity and eventual food availability. This has been proposed as partial support for reduced nesting abundance for loggerhead sea turtles in Japan. A finding that could have broader implications for other populations in the future if individuals do not shift feeding habitat (Chaloupka et al. 2008b). Pike (2014) estimated that loggerhead populations in tropical areas produce about 30 percent fewer hatchlings than do populations in temperate areas. Historical climactic patterns have been attributed to the decline in loggerhead nesting in Florida, but evidence for this is tenuous (Reina et al. 2013).

Tissues taken from loggerheads sometimes contain very high levels of organochlorines (Rybitski et al. 1995, McKenzie et al. 1999, Corsolini et al. 2000, Gardner et al. 2003, Keller et al. 2004a, Keller et al. 2004b, Keller et al. 2005, Alava et al. 2006, Perugini et al. 2006, Storelli et al. 2007, Monagas et al. 2008, Oros et al. 2009, Guerranti et al. 2013). High levels of organochlorines potentially suppress the immune system of loggerhead sea turtles and may affect metabolic regulation (Keller et al. 2004b, Keller et al. 2006, Oros et al. 2009). Organochlorine contaminants have the potential to depress immune function of loggerhead sea turtles (Keller et al. 2006) and likely have similar effects on other sea turtle species. These contaminants potentially cause deficiencies in endocrine, developmental, and reproductive health (Storelli et al. 2007).

Heavy metals, including arsenic, barium, cadmium, chromium, iron, lead, nickel, selenium, silver, copper, zinc, and manganese, have also been found in a variety of tissues in levels that increase with turtle size (Godley et al. 1999, Saeki et al. 2000, Anan et al. 2001, Fujihara et al. 2003, Gardner et al. 2006, Storelli et al. 2008, Garcia-Fernandez et al. 2009). These metals are likely accumulated from plants (Anan et al. 2001, Celik et al. 2006, Talavera-Saenz et al. 2007).

The omnivorous nature of loggerheads results in greater exposures to toxicants that biomagnify in the food web relative to other sea turtle species (Godley et al. 1999, McKenzie et al. 1999). Loggerhead sea turtles have higher mercury levels than any other sea turtle studied, but concentrations are an order of magnitude less than many toothed whales (Godley et al. 1999, Pugh and Becker 2001). Elevated mercury levels are associated with deformities in hatchlings versus healthy individuals (Trocini 2013). Arsenic occurs at levels several fold more concentrated in loggerhead sea turtles than marine mammals or seabirds. Antimicrobial agents in the marine environment are also of concern. Antibiotic-resistant bacteria found in loggerhead sea turtles suggested use and discharge of antimicrobial agents marine ecosystems (Foti et al. 2009).

Designated critical habitat. On July 10, 2014, NMFS and USFWS designated critical habitat for loggerhead sea turtles along the U.S. Atlantic and Gulf of Mexico coasts from North Carolina to Mississippi. While not within NMFS' jurisdiction, the USFWS designated about 1,102 km of coastal beach habitat as important for the recovery of the threatened Northwest Atlantic Ocean population of loggerhead sea turtles. The terrestrial designated critical habitat areas include 88 nesting beaches in coastal counties located in North Carolina, South Carolina, Georgia, Florida, Alabama, and Mississippi. These beaches account for 48 percent of an estimated 3965 km of coastal beach shoreline and about 84 percent of the documented nesting (numbers of nests) within these six states. Figure 3 shows the extent of designated critical habitat in Florida waters. These areas contain one or a combination of the following:

- Suitable nesting beach habitat.
- Sand suitable for nest construction and embryo development.

- Suitable nesting habitat with sufficient darkness so as not to deter nesting turtles
- Natural coastal processes or artificially created or maintained habit mimicking natural conditions.

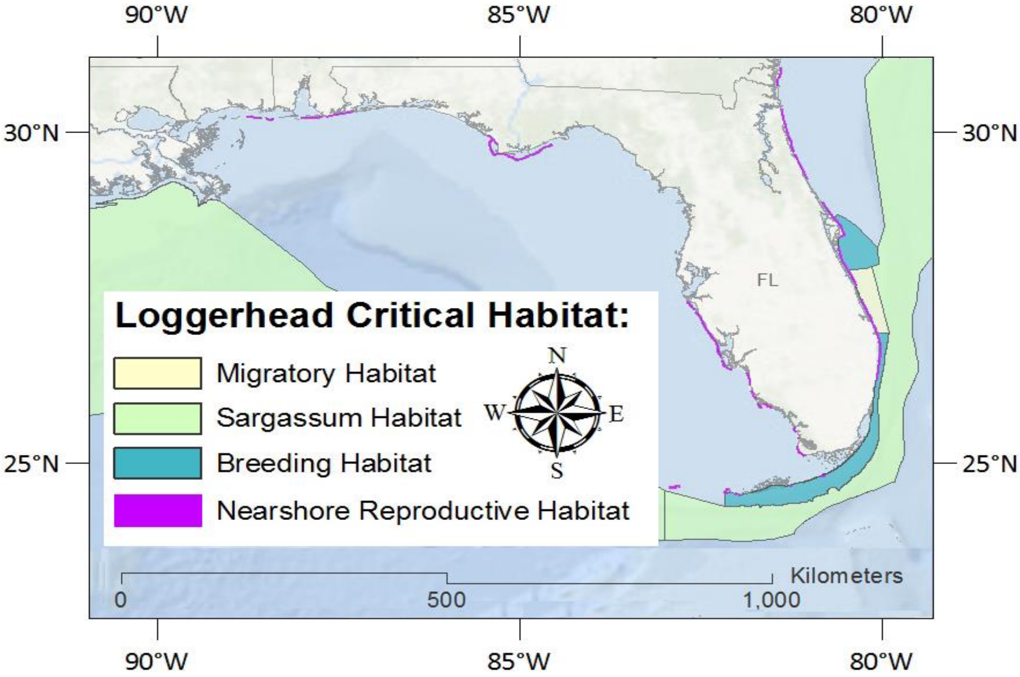

Figure 3. Loggerhead Sea Turtle Designated Critical Habitat in Florida.

6.3 Fish

6.3.1 Smalltooth Sawfish

Description. Although they are rays, sawfish physically more resemble sharks, with only the trunk and especially the head ventrally flattened. Smalltooth sawfish are characterized by their "saw," a long, narrow, flattened rostral blade with a series of transverse teeth along either edge. Adults weight 70 pounds (350 kg) and measure 18-25 feet (5.5-7 m) in length. Recent data from smalltooth sawfish caught off Florida indicate that young are born at 76 to 87 cm (Simpfendorfer and Wiley 2004). Males reach maturity at approximately 2.7 m and females at approximately 3.6 m (Simpfendorfer 2002). They live for 25-30 years and are "ovoviviparous," meaning the mother holds the eggs inside of her until the young are ready to be born.

Status. The U.S. smalltooth sawfish DPS was listed as endangered under the ESA on April 1, 2003. The smalltooth sawfish was common throughout their historic range up until the middle of the 20th century. The dramatic decline is attributed to the vulnerability of the sawfish life history to the impacts of fishing (both as bycatch and direct harvest) and habitat modification. As of 2001 the estimated U.S. population size was less than 5 percent of its size at the time of European settlement (Simpfendorfer 2001). The capture of a smalltooth sawfish off Georgia in 2002 is the first record north of Florida since 1963. This information and recent encounters in new areas beyond the core abundance area suggest that the population may be increasing. The abundance of juveniles encountered, including very small individuals, suggests that the population remains reproductively active and viable (Seitz and Poulakis 2002, Simpfendorfer 2003, Simpfendorfer and Wiley 2004). From 1989-2004, smalltooth sawfish relative abundance in the Everglades National Park has increased by about 5 percent per year (Carlson et al. 2007). Recent data from the International Sawfish Encounter Database suggest increasing trends in reported encounters of juvenile sawfish in Florida with a lag in increase in larger juveniles (Figure 4). The International Sawfish Encounter Database encounters are not systematic surveys, so reports in areas where people occur are over represented. Recovery of the species is expected to be slow given the species' life history and other remaining threats to the species, and therefore the population's future remains tenuous.

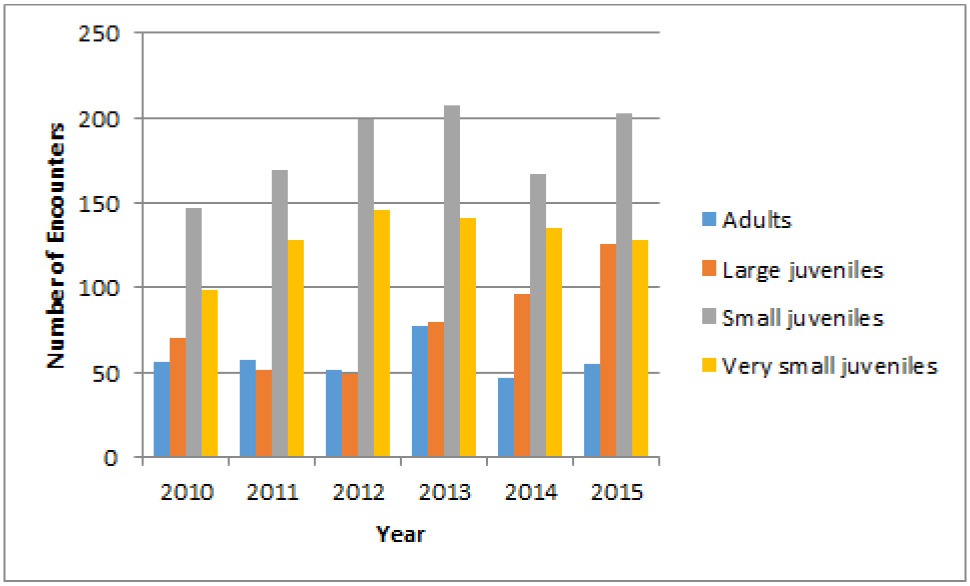

Figure 4. Smalltooth Sawfish Encounter Data within Florida Waters from the International Sawfish Encounter Database.

Use of Florida Waters. The majority of smalltooth sawfish encounters today are from the southwest coast of Florida between the Caloosahatchee River and Florida Bay. Outside of this core area, the smalltooth sawfish appears more common on the west coast of Florida and in the Florida Keys than on the east coast, and occurrences decrease the greater the distance from the core area (Simpfendorfer and Wiley 2004). Water temperatures no lower than 61°F to 64.4°F and the availability of appropriate coastal habitat serve as the major environmental constraints limiting the northern movements of smalltooth sawfish in the western North Atlantic (Simpfendorfer 2001). As a result, most records of this species from areas north of Florida occur during spring and summer periods (May to August) when inshore waters reach appropriately high temperatures. The data also suggest that smalltooth sawfish may use warm water outflows of power stations as thermal refuges during colder months, either to enhance their survival or because they have become trapped by surrounding cold water from which they would normally migrate. Almost all occurrences of smalltooth sawfish in warm-water outflows were during the coldest part of the year.

Habitat and Forage. Smalltooth sawfish occur in waters with a broad range of salinities from freshwater to full seawater (Simpfendorfer 2001). Poulakis and Seitz (2004) reported that almost all of the sawfish <3 m in length were found in water less than 10 m deep and 46 percent of encounters individuals >3 m in Florida Bay and the Florida Keys were reported at depths between 70 to 122 m. Recent data from sawfish encounter reports and from satellite tagging indicate mature animals occur regularly in waters in excess of 164 feet (Poulakis and Seitz 2004, Simpfendorfer and Wiley 2004). Since large animals are also observed in very shallow waters, it is believed that smaller (younger) animals are restricted to shallow waters, while large animals roam over a much larger depth range (Simpfendorfer 2001).

Smalltooth sawfish are most common in shallow coastal waters less than 25 m (Bigelow and Schroeder 1953). Nursery areas occur throughout Florida in areas of shallow water, close to shore and typically associated with mangroves (Simpfendorfer and Wiley 2004). Younger, smaller individuals tend to inhabit very shallow mud banks that are less than 1 foot (30 cm) deep and tides are a major factor in their movement (Simpfendorfer et al. 2010). As they grow, juveniles tend to occupy deeper habitat, but shallow areas (<1 m depth) remain preferred habitat (Simpfendorfer et al. 2010). Simpfendorfer (2003) investigated the home range size of very small, young-of-year (i.e., fish born within the last year <100 cm, n=2) and larger juvenile smalltooth sawfish (approximately 150 cm, n=2). The daily home ranges of the larger sawfish ranged from < 0.001 to 0.35 square kilometers. The data indicated a total home range of 0.12 and 1.22 square kilometers with a high level of site fidelity. For these larger individuals, there was less overlap in home range use between days, relative to smaller sawfish. Smaller young-of-year fish daily home ranges ranged from <0.001 to 0.007 square kilometers with overall home ranges of 0.01 and 0.08 square kilometers (see table 3 in Simpfendorfer 2003). In later work, Simpfendorfer et al. (2011) reported smalltooth sawfish in the nursery areas to have mean daily activity space of about 100-1000 m^2.

Smalltooth sawfish feed primarily on fish, with mullet, jacks, and ladyfish believed to be their primary food resources (Simpfendorfer 2001). In addition to fish, smalltooth sawfish also prey on shrimp and crabs, which are located by disturbing bottom sediment with their saw (Norman and Fraser 1937, Bigelow and Schroeder 1953).

Threats. The primary natural threat to smalltooth sawfish survival is the species low reproductive rate. In the face of reduced population sizes, this biological parameter means that recovery, at best, will be slow, and that catastrophic perturbations can have severer consequences to recovery. Historical decline has been largely due to fisheries interactions (see NMFS 2009 for a review). However, additional anthropogenic impacts result from habitat loss. Destruction of mangrove habitat, dredging, trawling and filling, and loss of reef habitat have negative impacts on all life stages of smalltooth sawfish. Habitat degradation due to runoff containing pesticides, eutrophying agents, and other contaminants can also have a negative impact on smalltooth sawfish habitat.

Designated critical habitat. On September 2, 2009, designated critical habitat was designated for smalltooth sawfish along the central and southwest coast of Florida (Figure 5). Mangrove and adjacent shallow euryhaline habitat are important nursery habitat for smalltooth sawfish. Nursery habitat consisting of areas adjacent to red mangroves and euryhaline habitats less than 0.9 m deep in southwestern Florida were later determined to be particularly significant (Norton et al. 2012). These areas are not within the waters affected by the estuary criteria covered in this opinion.

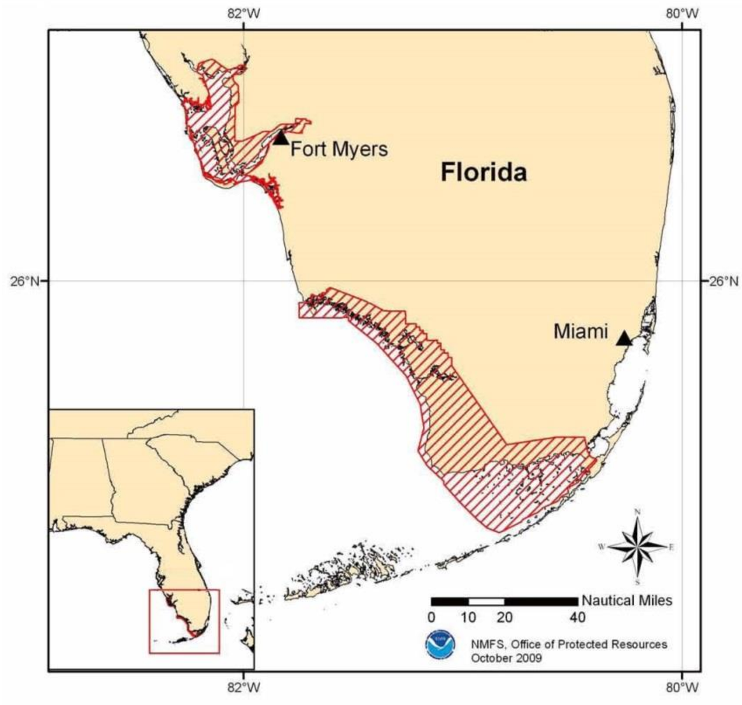

Figure 5. Smalltooth Sawfish Designated Critical Habitat.

6.3.2 Shortnose Sturgeon

Description. Adult shortnose sturgeon grow to up to 50 pounds (23 kg) and 4.5 feet (1.4 m) long. They have bony plates called "scutes" along their back. Lifespans average 30 years, but the species has been reported to live up to 67 years.

Status. Shortnose sturgeon were listed as endangered on March 11, 1967 under the Endangered Species Preservation Act and remained on the endangered species list with enactment of the ESA of 1973, as amended. NMFS’ recovery plan recognized 19 wild populations based on their strong fidelity to their natal streams, and captive populations maintained for educational and research purposes (NMFS 1998). Despite their longevity, the viability of sturgeon populations is sensitive to variability in juvenile recruitment and survival (Anders et al. 2002, Gross et al. 2002, Secor et al. 2002).

The largest shortnose sturgeon population inhabits Hudson River and appears to have increased to approximately 60,000 individuals (NMFS 1998). The Kennebec River population also appears to be increasing. The most recent estimate of 9,500 fish (Squiers 2003), suggests a 30 percent increase over approximately twenty years. The Delaware River population appears to be stable (Hastings et al. 1987, O'Herron II et al. 1993, Brundage III 2006). Populations are relatively small south of Chesapeake Bay, with the largest Altamaha River population about an order of magnitude smaller than the Hudson River population and the Ogeechee River population, which appears to be declining, is orders of magnitude smaller than the Hudson River population (NMFS 1998, DeVries 2006). By some accounts, shortnose sturgeon, populations may be extinct in several basins (e.g., St. Johns in Florida, St. Marys, Potomac, Housatonic, and Neuse rivers). Those few fish that have been observed in these basins are generally presumed to be immigrants from neighboring basins.

Use of Florida Waters. Rogers and Weber (1995), Kahnle et al. (1998a), and Collins et al. (2000) concluded that shortnose sturgeon are extinct from the St. Johns River in Florida and the St. Marys River along the Florida and Georgia border. However, a 2010 report from the shortnose sturgeon status review team indicated both Atlantic and shortnose sturgeon are found in the St. Marys River. A 2001-2004 Florida FFWCC shortnose sturgeon survey in the St. Johns yielded a single individual (63.5 cm total length; 1,589, FFWCC 2007c). This survey applied the NMFS survey protocol and at that time it was realized that this protocol may need modification for use within the St. John system given the broad river coupled with fast moving water. Applying a revised protocol may either confirm the original observations or reveal a larger population (SSSRT 2010). No reproduction of sturgeon in the St. Johns River has ever been documented, and no large adults have been positively identified. Given the marginal spawning habitat, it is possible that shortnose sturgeon never actively spawned in the St. Johns. The species is retained in this analysis because the St. Marys and St. Johns Rivers may yet contain populations and these rivers may eventually serve the species in future recovery.

Habitat and Forage. Habitat use in fresh water during summer and winter months overlaps between adult and age-1 shortnose sturgeon (O'Herron II et al. 1993, Rogers and Weber 1995, Kynard et al. 2000). Kynard et al. (2000) found that both age classes preferred deep-water curves with sand and cobble to higher velocity runs during winter months and shifted to channel habitat as water temperatures rose in summer months. Many fish also exhibited diel movement patterns between deeper waters during the day and shallower waters at night (Kynard et al. 2000). During the summer, at the southern end of their range the species tends to congregate in cool, deep, areas of rivers (Flournoy et al. 1992, Rogers and Weber 1995, Weber 1996).

Shortnose sturgeon have ventrally located, sucker-like mouths, structured for feeding on benthos. Foraging generally occurs in areas with abundant macrophytes, where juvenile and adult shortnose sturgeon feed on amphipods, polychaetes, and gastropods (Dadswell et al. 1984, Moser and Ross 1995, NMFS 1998). Starting as larvae, sturgeon use electroreception to identify prey. Olfaction and taste are also likely important to foraging, while vision is thought to play a minor role (Miller 2004). As adults, a significant portion of the shortnose sturgeon diet may consist of freshwater mollusks (Dadswell et al. 1984). Based on observations by Kynard et al. (2000), shortnose sturgeon will consume the entire mollusk, excreting the shell after ingestion.

Threats. Yellow perch, sharks, and seals are predators of shortnose sturgeon juveniles (NMFS 1998). Shortnose sturgeon have declined from the combined effects from the construction of hydropower and water diversion projects, dredging and blasting, water pollution, fisheries, and hatcheries. The construction of dams has resulted in substantial loss of shortnose sturgeon habitat and access to spawning areas along the Atlantic seaboard. The effects of fishing in the late nineteenth and early twentieth centuries may have latent and long-lasting impacts on some populations (NMFS 1998).

Studies demonstrate that shortnose sturgeon carry a wide number of potentially hazardous contaminants. Individuals from the Delaware River contain heavy metals, dioxins, dibenzofurans, polychlorinated biphenyls, dichlorodiphenyltrichloroethane degradates, bis (2-ethylhexyl) phthalate, di-n-butylphthalate, and chlordane (ERC 2002). Most of these metals, dioxins, deibenzofurans, and polychlorinated biphenyls were also found in shortnose sturgeon in the Kennebec River (ERC 2003).

Climate change has the potential to affect sturgeon through disruption of spawning habitat, barriers to migration, and degraded water quality. Increased extremes in river flow can disrupt and fill in spawning habitat that sturgeon rely upon (ISAB 2007). Low flow rates during migration can impede or block sturgeon movement. Sturgeon are directly sensitive to elevated water temperatures. Increased mortality can occur if cooler water refuges are not available in freshwater habitats. If temperatures rise beyond thermal limits for extended periods, the species range can contract, as southern habitats may become uninhabitable (Lassalle et al. 2010). Apart from direct changes to sturgeon survival, altered water temperatures may also disrupt the availability of prey or result in increased water withdrawals to support agriculture (ISAB 2007).

Designated critical habitat. Designated critical habitat has not been established for shortnose sturgeon.

6.3.3 South Atlantic Distinct Population Segment of the Atlantic sturgeon

Description. Atlantic sturgeon are a long-lived, late maturing, iteroparous, anadromous species. They are bluish-black or olive brown with paler sides and a white belly. This species is a bottom-feeder that has a ventral suctorial mouth without teeth, four whiskers halfway between the snout and mouth, five rows of scutes (armor-like scales), and a tail longer on top than on the bottom. They grow to up to 800 pounds (370 kg) and 14 feet (4.3 m) long and the average lifespan is 60 years.

Status. NMFS listed five DPSs of Atlantic sturgeon: the New York Bight, Chesapeake Bay, Carolina, and South Atlantic DPSs as endangered and the Gulf of Maine DPS as threatened on February 6th 2012. The Interim 4(d) Rule for Protective Regulations for the Gulf of Maine DPS was published November 11, 2013.

Atlantic sturgeon were once present in 38 river systems and, of these, spawned in 35 of them.

Individuals are currently present in 36 rivers, and spawning occurs in at least 20 of these. Record landings were reported in 1890, where over 3350 metric tons of Atlantic sturgeon were landed from coastal rivers along the Atlantic Coast (Smith and Clugston 1997, Matthiopoulos and Aarts 2010). Between 1890 and 1905, Atlantic sturgeon populations declined dramatically due to sale of meat and caviar. The fishery collapsed in 1901 and was closed by the Atlantic States Marine Fisheries Commission in 1998, when a coastwide fishing moratorium was imposed for 20 to 40 years, or at least until 20 year classes of mature female Atlantic sturgeon were present (ASMFC 1998). The Hudson River (New York Bight DPS) and Altamaha River (South Atlantic DPS) are the two largest spawning populations on the East Coast. Kahnle et al. (1998b) reported that approximately 870 adults per year returned to the Hudson River between 1985 and 1995. Peterson et al. (2008) reported that approximately 324 and 386 adults per year returned to the Altamaha River in 2004 and 2005, respectively. Other DPSs within the U.S. are predicted to have fewer than 300 adults spawning per year. However, evaluating the status of the species depends on the status of the smaller extant populations because maintaining those populations maintains genetic heterogeneity and having a broad range prevents a single catastrophic event from causing their extinction.

Habitat, lifecycle, and forage. Modern genetic analyses suggest that, despite extensive mixing in coastal waters, Atlantic sturgeon exhibit high fidelity to their natal rivers (Harwood 2010) and most rivers appear to support independent populations (Waldman and Wirgin 1998, Wirgin et al. 2000, King et al. 2001, Wirgin et al. 2002, Grunwald et al. 2008)). Sub-adult and adult Atlantic sturgeon spend most of their life in the marine environment. Migratory sub-adults and adults normally occur in shallow (10-50m) waters dominated by gravel and sand substrate (Stein et al. 2004b). Spawning adults generally migrate upriver in the spring and early summer; this includes February-March in southern systems, April-May in mid-Atlantic systems, and May-July in Canadian systems (Smith 1985, Bain 1997, Smith and Clugston 1997) (Kahnle et al. 1998b).

Sturgeon larvae assume a bottom-dwelling existence until the yolk sac is absorbed then move downstream to rearing grounds (Kynard and Horgan 2002) using benthic structure (e.g., stream bed gravel matrix) as day-time refugia (Kynard and Horgan 2002). Juvenile sturgeon continue to move further downstream into brackish waters, and eventually become residents in estuarine waters for months or years. Estuaries along the coast that do not support Atlantic sturgeon spawning populations may still be important rearing, feeding, and migration habitats (Harrison and Thurley 1974, Dovel and Berggren 1983, Bain 1997).

Atlantic sturgeon feed primarily on polychaetes, isopods, and amphipods in the marine environment, while in fresh water, they feed on oligochaetes, gammarids, mollusks, insects, and chironomids (Moser and Ross 1995, Johnson et al. 1997, Haley 1998, Haley 1999, Brosse et al. 2002, Guilbard et al. 2007, Savoy 2007, Collins et al. 2008). There is conflicting evidence whether fish forage in the springtime or in freshwater (Brosse et al. 2002, Collins et al. 2008).

Water quality conditions required by the species were reviewed in (Schlenger et al. 2013). Minimum water quality requirements for young-of-year sturgeon include water temperatures of

0-28°C, salinity of 0-22 ppt, and DO concentrations of at least 3.3 mg/L. Optimal conditions for young-of-year fish are water temperatures of 16-24 °C, 3.5 to 18.5 ppt salinity, and DO concentration of 5 mg/L. Yearlings differ from young-of-year fish only in their salinity tolerance and optima, with salinities of 0-29 ppt required and 18.5-25.5 ppt salinity optimal.

Use of Florida Waters. Atlantic sturgeon were abundant in the St. Marys and St. Johns Rivers prior to 1884 (Hamlen 1884). Atlantic sturgeon were once thought to be extirpated in the St. Marys River. Recent captures of sub-adult sturgeon suggest the potential for regaining breeding populations in Florida. The FFWCC's 2011 Biological Status Review reported these captures:

> *In January 2010, shrimp try-nets in 15 meter depths were used for chase-trawling chilled sea turtles during Kings Bay Trident submarine channel maintenance. During this exercise, a trawler netted and released 21 sub-adult (~1 meter) Atlantic sturgeon in the St. Marys estuary (Slay, Pers. Comm. 2010). Dr. Doug Peterson's University of Georgia sampling study also captured nine subadult (~1 meter) Atlantic sturgeon in the tidally-influenced St. Marys, ranging through summer, fall, and winter captures during 2010 (Peterson, Pers. Comm. 2010). In February of 2011, two year-one/year-two juvenile (~40 centimeter) Atlantic sturgeon were caught on hook and line, from the shore, in the St. Johns River (Snyder, Pers. Comm. 2011). This could suggest that the nearby Atlantic sturgeon populations are increasing sufficiently to re-establish resident juvenile populations in the St. Marys and St. Johns Rivers. This is the first step, which necessarily precedes the St. Marys River and St. Johns River regaining their own breeding populations, as the resident juveniles mature. So the status is "extirpated or nearly extirpated, but migrants are occupying northeast Florida rivers."*

During time of criteria development (2012-2013), Florida sought advice from NMFS Southeast Region on the use of Florida waters by ESA-listed sturgeon. NMFS Southeast Region reported that most of the sturgeon captures in the St. Marys occurred between river km 26 and 44 and that there is no evidence that spawning has occurred in the St. Marys River due to unfavorable natural conditions. Sturgeon were captured in portions of the river having limited anthropogenic inputs, but at DO levels as low as 2.7 mg/L. Florida attributes low DO levels in the St. Marys to natural wetland inputs, contributions of organic matter from bankside vegetation, and low flows.

The map in Figure 6 identifies the distribution and sites of capture for Atlantic sturgeon in Florida up to 2013. The St. Marys River accounts for 30 of the 34 recorded Atlantic sturgeon captures in Florida. Captures have also been reported in the Nassau (1 capture) and St. Johns Rivers (3 captures), but these are considered vagrant occurrences (FFWCC 2013). More recently, the June 3, 2016, NMFS designated critical habitat for the South Atlantic DPS of the Atlantic sturgeon to include the St. Marys River. The St. Marys was identified as a spawning river for Atlantic sturgeon based on the capture of young-of-year Atlantic sturgeon. Nine Atlantic sturgeon were captured in sampling efforts between May 19 and June 9, 2014. Captured fish ranged in size from 293 mm (young-of-year) to 932 mm (subadult). This is a possible indication of a slow and protracted recovery in the St. Marys (see 81 FR 36077). Meanwhile the designated critical habitat did not include the St. Johns River because it does not appear to support spawning and juvenile recruitment or contain suitable habitat features to support spawning because spawning areas are inaccessible due to man-made structures and alterations. In the southeast U.S., Atlantic sturgeon appear to spawn in the fall (Personal Communication from Jason Kahn 2015).

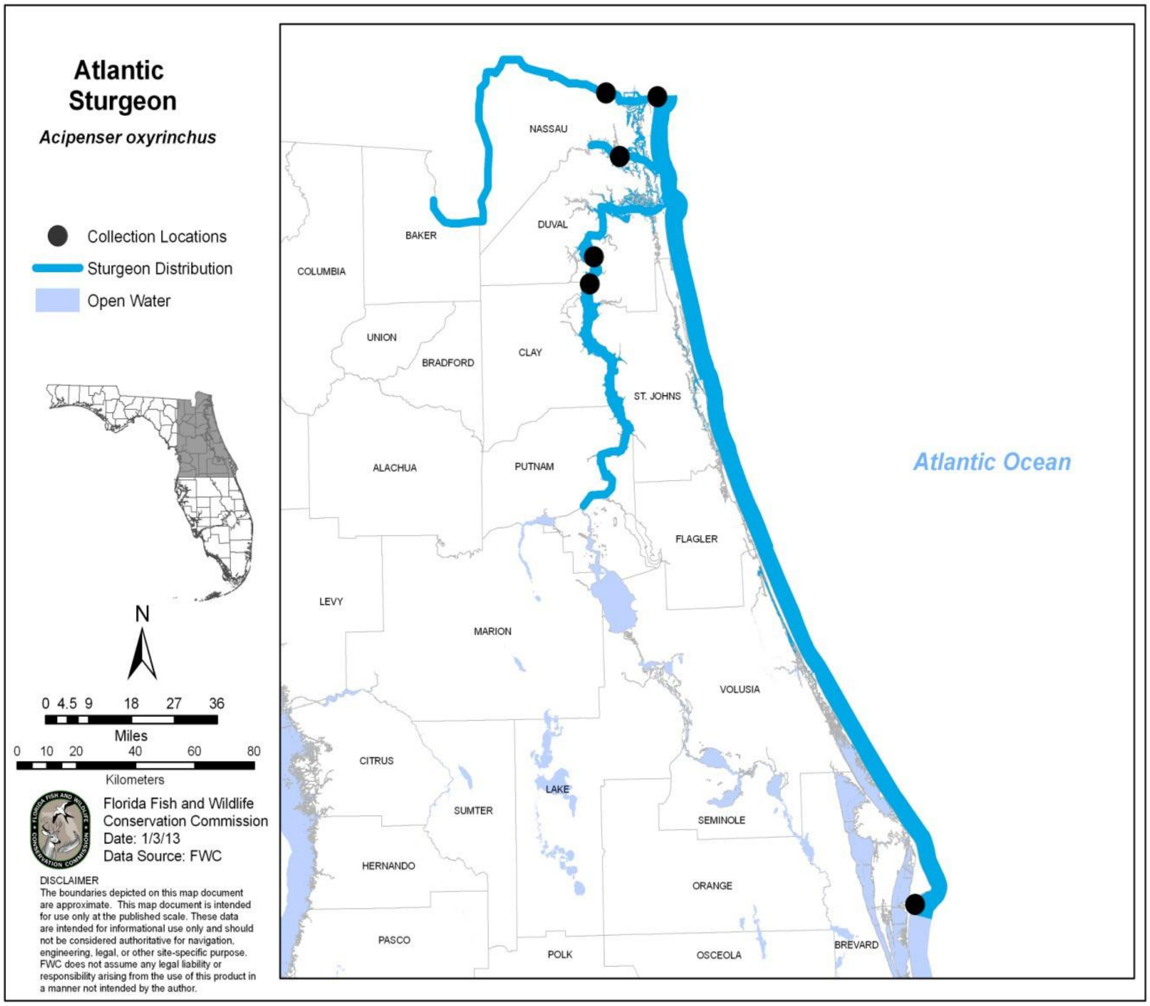

Figure 6. Florida Priority Watershed Areas Known or Having Potential to Harbor Atlantic Sturgeon.

Threats. Alee effects, the phenomenon of declining individual fitness in sparse populations, may be influencing small populations in some rivers. Water quality, ship strikes, bycatch, dams, and poaching all contribute to currently depressed populations of Atlantic sturgeon.

The 2011 biological status review report (FFWCC 2011) placed a priority on habitat management actions that restore minimum DO concentrations exceeding 3.0 ppm throughout river systems.

In large river systems like the Delaware, James and Hudson rivers, large ships move upstream from the mouths of the river to ports upstream through narrow shipping channels. The channels are dredged to the approximate depth of the ships, usually leaving less than 6 feet of clearance between the bottom of ships and the benthos of the river. Because of the size of the propellers used on large ships, everything along the bottom is sucked through the propellers. The act of dredging channels can also kill sturgeon. Dredging projects in the Kennebec, Delaware, James, Cape Fear, and Savannah Rivers put Atlantic sturgeon at moderate risk (ASSRT 2007). Dredging primarily affects sturgeon by removing food resources and homogenizing habitat, eliminating holding areas and other high quality habitat.

Atlantic sturgeon are caught as bycatch in several fisheries both within river systems and along the coast. In the James River, bycatch in the striped bass fishery poses a moderately high risk to the species, while it poses a moderate risk in nearly every other river system on the East Coast (ASSRT 2007). While these determinations were made for Atlantic sturgeon in each river system, the majority of the commercial fisheries interactions occur in estuaries and along the coast, where sturgeon from all rivers could be captured as bycatch.

On the East Coast, there is no good means of fish passage for Atlantic sturgeon in the systems with dams. Sturgeon in the Santee-Cooper River system and the Cape Fear River are at a moderately high risk because of dams. Additionally, sturgeon in the Neuse River are at a moderate risk from dams.

Industrialization, poor water quality, and loss of habitat adversely affect Atlantic sturgeon populations (Van Eenennaam et al. 1996, Jager et al. 2001, Collins et al. 2002, Stein et al. 2004a). Most Atlantic sturgeon managers and researchers consider water quality as a moderate risk to every DPS in the United States (ASSRT 2007). Atlantic sturgeon are sensitive to pesticides, heavy metals, and other toxins in the aquatic environment.

Designated critical habitat. Designated critical habitat has not been proposed for Atlantic sturgeon.

6.3.4 Gulf Sturgeon

EPA’s approval of the Florida NNC was evaluated for effects on the Gulf sturgeon by the USFWS. The jurisdictional disposition for ESA section 7 consultations for the Gulf sturgeon was clarified in the critical habitat designation. The USFWS is responsible for all consultations regarding Gulf sturgeon and designated critical habitat in all riverine actions and in those estuarine actions for which the EPA is the action agency. NMFS is responsible for all consultations regarding Gulf sturgeon and designated critical habitat in marine waters. Federal projects that extend into the jurisdiction of both the Services are to be consulted on by the USFWS with internal coordination with NMFS. Therefore, consultation on Gulf sturgeon for this action is the responsibility of USFWS.For this reason, Gulf sturgeon are not considered further in this opinion.

6.3.5 Nassau grouper

Description. The Nassau grouper is a long-lived, moderate sized marine fish with large eyes and a robust body. The range of color is wide, but ground color is generally buff, with 5 dark brown vertical bars and a large black saddle blotch on top of caudal peduncle and a row of black spots below and behind eye. Color pattern can change within minutes from almost white to bicolored to uniformly dark brown, according to the behavioral state of the fish (Longley 1917, Colin 1992, Heemstra and Randall 1993, Carter et al. 1994).

Status. The Nassau grouper has been designated a candidate species since 1991. NMFS began a status review on the species in 1993 and identified research that needed to be conducted to fill some of the gaps in the information concerning the species biology, genetics and habitat requirements. Under the authority of the Magnuson-Stevens Fishery Conservation and Management Act, NMFS classified the Nassau grouper as “overfished” in its October 1998 “Report to Congress on the status of Fisheries and Identification of overfished Stocks.” The species was proposed for listing as a threatened species under the ESA September 2, 2014. The final listing for this species was published on June 29, 2016 to become effective July 29, 2016.

Habitat, lifecycle, and forage. The Nassau grouper is primarily a shallow-water, insular fish species found from inshore to about 330 feet (100m) depth. The species is considered a reef fish, but it transitions through a series of developmental shifts in habitat. As larvae, they are planktonic. After an average of 35-40 days and at an average size of 32 mm total length, larvae recruit from an oceanic environment into demersal habitats (Colin 1992, Eggleston 1995). Following settlement, Nassau grouper juveniles are reported to inhabit macroalgae (primarily Laurencia spp.), coral clumps (*Porites* spp.), and seagrass beds (Eggleston 1995, Dahlgren 1998). Juvenile Nassau grouper (120-150 mm total length) are relatively solitary and remain in specific areas for months (Bardach 1958). Juveniles of this size class are associated with macroalgae, and both natural and artificial reef structure. As juveniles grow, they move progressively to deeper areas and offshore reefs (Colin et al. 1997, Brill et al. 2008). Schools of 30-40 juveniles (250-350 mm total length) were observed at 8-10 m depths in the Cayman Islands (Brill et al. 2008). No clear distinction can be made between types of adult and juvenile habitats, although a general size segregation with depth occurs—with smaller Nassau grouper in shallow inshore waters (3 to 16 m) and larger individuals more common on deeper (17 to 55 m) offshore banks (NMFS 2013a). Adult Nassau grouper tend to be relatively sedentary and are generally associated with high relief coral reefs or rocky substrate in clear waters to depths of 130 m.

Maximum age has been estimated up to 29 years, based on an ageing study using sagittal otoliths (Belcher and Jennings 2010). Most studies also indicate rapid growth, which has been estimated to be about 10 mm/month total length for small juveniles, and 8.4 to 11.7 mm/month for larger juveniles (30-270 mm total ength; Beets and Hixon 1994, (Eggleston 1995). Maximum size is about 122 cm total length and maximum weight is about 25 kg (Heemstra and Randall 1993, Humann and DeLoach 2002). Generation time (the average age of parents in the population) is estimated as 9-10 years (Sadovy and Colin 1995). Nassau grouper reproduce in site specific spawning aggregations. Spawning aggregations, of a few dozen up to perhaps thousands of individuals have been reported from the Bahamas, Jamaica, Cayman Islands, Belize, and the Virgin Islands. These aggregations occur in-depths of 20-40 mat specific locations of the outer reef shelf edge. Spawning takes place in December and January, around the time of the full moon, in waters 25-26 degrees C.

Use of Florida Waters. The species is distributed throughout the islands of the western Atlantic including Bermuda, the Bahamas, southern Florida, and along the coasts of central and northern South America. It is not known from the U. S. Gulf of Mexico except at Campeche Bank off the coast of the Yucatan Peninsula, at Tortugas, and off Key West. Adults are generally found near coral reefs and rocky bottoms while juveniles are found in shallower waters in and around coral clumps covered with macroalgae (*Laurencia* spp.) and over seagrass beds. Their diet is mostly fishes and crabs, with diet varying by age/size. Juveniles feed mostly on crustaceans, while adults (>30 cm; 11.8 in) forage mainly on fish. The Nassau grouper usually forages alone and is not a specialized forager.

Threats. Because Nassau grouper spawn in aggregations at historic areas and at very specific times, they are easily targeted during reproduction. Because Nassau grouper mature relatively late (4-8 years), many juveniles may be taken by the fishery before they have a chance to reproduce.

Designated critical habitat. Critical habitat is not designated for species proposed for listing as endangered or threatened under the ESA.

6.4 Corals

Seven species of hard corals that occur within Florida waters are listed as threatened under the ESA: elkhorn coral, staghorn coral, lobed star coral, boulder star coral, mountainous star coral[4], pillar coral, and rough cactus coral. Elkhorn and staghorn corals were listed together as threatened under the ESA on May 9, 2006. The remaining species were listed as threatened on September 10, 2014.

Hard corals are colonies of small animals with calcium carbonate skeletons that collaboratively form reefs by creating structures that are firmly attached to the sea floor. Rapid calcification by hard corals is made possible by the symbiotic algae which reside within coral polyps, called zooxanthellae. Reef-building corals do not thrive outside of an area characterized by a fairly narrow mean temperature range (typically 25 °C-30 °C). Soft corals differ from hard corals in that they are flexible, have calcareous particles in their body walls for structural support, can be found in both tropical and cold ocean waters, do not grow in colonies or build reefs, and do not always contain zooxanthellae.

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[4] This species presence is based on a strong prediction of occurrence and not confirmed record (Veron 2014).

Zooxanthellae photosynthesize during the daytime producing energy for the host coral. At night, polyps extend their tentacles to filter-feed on microscopic particles in the water column such as zooplankton, providing additional nutrients for the host coral. In this way, reef-building corals obtain nutrients autotrophically (i.e., via photosynthesis) during the day, and heterotrophically (i.e., via predation) at night.

Most coral species use both sexual and asexual propagation. Asexual reproduction most commonly involves fragmentation, where colony pieces or fragments are dislodged from larger colonies to establish new colonies, although the budding of new polyps within a colony can also be considered asexual reproduction. In many species of branching corals, fragmentation is a common and sometimes dominant means of propagation.

Biological and physical factors affect spatial and temporal patterns of recruitment. These include substrate availability and community structure, grazing pressure, fecundity, mode and timing of reproduction, behavior of larvae, hurricane disturbance, physical oceanography, the structure of established coral assemblages, and chemical cues (Lewis 1974, Birkeland 1977, Goreau et al. 1981, Rogers et al. 1984, Baggett and Bright 1985, Harriott 1985, Hughes and Jackson 1985, Sammarco 1985, Morse et al. 1988, Fisk and Harriott 1990, Richmond and Hunter 1990).

Threats common to corals. The coral species living off the coast of Florida are vulnerable to the same anthropogenic stressors which threaten corals worldwide: climate change, fishing impacts, and pollution. The following discussion was adapted from the NOAA Coral Reef Conservation Program threat summaries (http://coralreef.noaa.gov/threats, July 13, 2015), the NOAA South East Region fisheries recovery outline for ESA-listed corals in the region, and the listing documentation for corals listed as threatened under the ESA.

Increased water temperatures and ocean acidification resulting from climate change increases coral vulnerability to infection or disease and bleaching and impairs the construction and maintenance of calcium carbonate-based skeletal frameworks. Ocean acidification is caused by increased dissolved CO_2 in ocean water. This changes the solubility and form of sea water minerals in even slightly more acidic sea water. Most critically, acidification reduces seawater saturation with aragonite, the form of calcium carbonate used by corals and other marine species to construct protective shells and skeletal frameworks, thereby eroding the shells which form coral hard parts (Anthony et al. 2008, De’ath et al. 2009, Wei et al. 2009, Crawley et al. 2010). Acidification also reduces thermal tolerance of corals, meaning that bleaching can occur at lower temperatures (Anthony et al. 2008).

Mass coral bleaching, which results from the expulsion of the symbiotic zooxanthellae algae, is linked to excursions in ocean temperatures outside of coral physiological tolerances. Warm water bleaching events typically co-occur with high subsurface light levels and are associated with major El Niño-Southern Oscillation events (e.g., 1982–83, Glynn and D'croz, 1990; 1997–98, Wilkinson et al 2000; and 2002, Berkelmans et al 2004) . Laboratory experiments have confirmed this association (Coles and Jokiel 1978, Glynn and D'croz 1990). Increased coral mortality due to the stress from bleaching alters reef habitats, structures, and biodiversity (Eakin

2001, Graham et al. 2006). The most severe and extensive Caribbean mass warm water bleaching event occurred in 2005. Only localized warm water bleaching was observed in the years between 2006 and 2013 (Manzello 2015) and a cold water bleaching event occurred in the Florida Keys over the winter of 2009-2010. In 2005, wide-scale bleaching occurred throughout the Caribbean with wide-scale mortality, with some areas reaching 95 percent of coral colonies affected (Wilkinson and Souter 2008). Puerto Rico and Florida also experienced disease rates of 50 percent of coral colonies or greater. Following the 2005 bleaching event, monitoring data indicate that total coral cover is now less than 12 percent on many reefs (Rogers et al. 2008). Coral mortality due to the 2005 bleaching event was more severe than at any time in the last 40 years of monitoring in U.S. Virgin Islands (Woody et al. 2008). Bleaching events can lead to increased thermal tolerance in affected reefs, meaning that subsequent bleaching events are not as severe (Maynard et al. 2008).

Taken together, disease and ocean warming are major threats affecting the potential for coral recovery in the southeast U.S. because they are severe, ongoing, synergistic, and have increased in the recent past. Mortality rates after disease and bleaching events have not been compensated for through recruitment or growth. Sea-surface temperature is expected to continue to rise over time and exacerbate disease impacts. Climate change effects will impact corals, such as sea level rise, altered ocean circulation, and changes in the frequency, intensity, and distribution of tropical storms. These changes may increase physical damage to coral reefs (Madin et al. 2012, Teixidó et al. 2013) or harm corals by severely reducing salinity with large influxes of stormwater runoff (Berkelmans et al. 2012, Lough et al. 2015). Hurricanes fueled by warmer waters can cause wide-scale inhibition of recruitment in years following storm passage as well as physical damage to coral colonies themselves (Mallela and Crabbe 2009). A record number of hurricanes in 2005 caused extensive damage to coral reefs; the prevalence of hurricanes and subsequent coral reef damage has been linked to climate change (Wilkinson and Souter 2008).

Fishing impacts on coral reefs include direct harvests of coral, cascading effects due to the removal or reduction of important functional species from coral reef communities, and physical damage by certain fishing gears and fishing methods that can directly contact coral reefs and the anchoring of fishing vessels on coral reefs. Cascading effects resulting from altered trophic structure of the reef community degrades coral condition and habitat and increases synergistic stress effects (e.g., bleaching, disease).

Coastal development contributes localized threats through run-off of land-based pollutants, including excess nutrients and sediment, and through physical damage from activities such as dredging, cable and pipeline deployment, construction, and beach nourishment. Suspended sediment and sediment deposition act to limit coral growth, feeding patterns, photosynthesis, recruitment, and survivorship. Reductions in long-term water clarity can also reduce the coral photosynthesis to respiration ratio. Telesnicki and Goldberg (1995) and Yentsch et al. (2002) found that elevated turbidity levels did not affect gross photosynthetic oxygen production, but did lead to increased respiration that consumed the products of photosynthesis with little remaining for coral growth. Excessive sedimentation can smother corals and increased nutrient availability promotes algal growth on corals, leading to light blockage to zooxanthellae and death of corals (ABRT 2005). Although reefs in the Florida Keys currently experience about 10 percent macroalgal cover or less, much of the wider Caribbean Sea may exceed 20 percent cover (Bruno 2008), inhibiting and reducing coral survival.

6.4.1 The Acropora: Elkhorn and Staghorn Coral

Description. Elkhorn coral forms frond-like branches radiating from a central trunk. Colonies can reach 6.6 feet high and 13 feet in diameter (Veron 2000). Corallites (branches) are tube-like and porous, 0.08 inch to 0.16 inch long, about 0.08 inch in diameter, white near the growing tip, and brown to tan away proximally. Staghorn coral branches are irregular, with secondary branches forming at 60 to 90 degree angles relative to a primary branch branches. Individual colonies are up to 5 feet across and typically form monospecific thickets. Branches are 0.1 inch to 0.6 inch in diameter and rarely may grow back together. Prominent axial corallites form at branch tips; bract-like corallites radiate symmetrically around branches. Tissue color ranges from golden yellow to medium brown, with little or no color near the growing branch tips.

Status. Precipitous declines for these species began in the early 1980s throughout their range. Although quantitative data on historical distribution and abundance are scarce, best available data indicate declines in abundance (coverage and colony numbers) by greater than 97 percent. Monitoring data do not indicate significant recovery after the widespread mortality associated with the 2005 bleaching event (Rothenberger et al. 2008, Woody et al. 2008). Overall, colonies of Atlantic *Acropora* have declined by up to 98 percent and live colonies were no longer present at many study sites in the U.S. Virgin Islands following the 2005-2006 bleaching event.

Both species occur in the Florida Keys, Abaco Island (The Bahamas), Alacran Reef, Mexico, Belize, Colombia, Costa Rica, Guatemala, Honduras, Nicaragua, Panama, Venezuela, Bonaire, Cayman Islands, Jamaica, Puerto Rico, U.S. Virgin Islands, Navassa, and throughout the West Indies (Goreau 1959, Kornicker and Boyd 1962, Storr 1964, Scatterday 1974, Jaap 1984, Dustan and Halas 1987, NMFS 2006). However, abundance within the distribution is reduced, largely due to water temperature and quality issues.

Growth and reproduction. Branching species, such as acroporid corals, grow differentially in response to light such that coral polyp growth maximizes exposure to available light (Kaniewska et al. 2009). The dominant mode of reproduction for elkhorn and staghorn corals is asexual fragmentation and dispersal (Tunnicliffe 1981, Bak and Criens 1982). Sexual reproduction is accomplished by releasing sperm and egg during spawning events. Colonies are referred to as simultaneous hermaphrodites, meaning that a given colony contains both female and male reproductive sex organs (Szmant 1986). Spawning events are relatively short, with gametes released only a few nights during July, August, and/or September. Once fertilization occurs, planktonic larvae form before settling and metamorphosizing on appropriate substrates, preferably coralline algae (Bak 1977, Sammarco 1980, Rylaarsdam 1983). Initial calcification ensues and develop into daughter corallites. Studies indicate that larger colonies (as measured by surface area of the live colony) have higher fertility and fecundity rates (Soong and Lang 1992).

Biological and physical factors affect spatial and temporal patterns of recruitment. These include substrate availability and community structure, grazing pressure, fecundity, mode and timing of reproduction, behavior of larvae, hurricane disturbance, physical oceanography, the structure of established coral assemblages, and chemical cues (Lewis 1974, Birkeland 1977, Goreau et al. 1981, Rogers et al. 1984, Baggett and Bright 1985, Harriott 1985, Hughes and Jackson 1985, Sammarco 1985, Morse et al. 1988, Fisk and Harriott 1990, Richmond and Hunter 1990). Larval recruitment is influenced by the type and availability of benthic substrate, with certain types of coral or rock substrates resulting in greater or lesser recruitment success (Ritson-Williams et al. 2009).

Habitat. Although staghorn coral colonies are sometimes found interspersed among colonies of elkhorn coral, they are generally in deeper water or seaward of the elkhorn zone and more protected from wave action. Staghorn coral occur in back reef (landward slope) and fore reef (seaward slope) environments from 0-100 feet (0 to 30 m) deep. The upper limit is defined by wave forces, and the lower limit is controlled by suspended sediments and light availability. Fore reef zones at intermediate depths of 15-80 feet (5-25 m) were formerly dominated by extensive single species stands of staghorn coral until the mid-1980s. In southeastern Florida, this species historically occurred on the outer reef (52 to 66 feet), on spur, groove bank, and transitional reefs, and on octocoral-dominated hard-bottom (Goldberg 1973, Davis 1982, Jaap 1984, Wheaton and Jaap 1988). Colonies were common in back- and patch-reef habitats (Gilmore and Hall 1976, Cairns 1982).

Colonies of elkhorn coral often grow in dense stands and form interlocking framework known as thickets in fringing and barrier reefs, ranging in-depth from 3.3 to 49 feet (Jaap 1984, Dustan 1985, Dustan and Halas 1987, Tomascik and Sander 1987b, Wheaton and Jaap 1988). However, optimal depth range is considered to be 3.3 to 16.4 feet in-depth, with possible exposure at low tide (Goreau and Wells 1967). Elkhorn coral thrive in shallow reef zones where wave energy is a significant factor. In areas with strong wave energy conditions only isolated colonies occur, while denser thickets may develop in intermediate wave energy conditions (Geister 1977). The preferred habitat of elkhorn coral is the seaward face of a reef (Shinn 1963, Cairns 1982, Rogers et al. 1982).

Threats. White band disease is thought to be the major factor responsible for the rapid loss of Atlantic *Acropora* due to mass mortalities. White band disease is the only coral disease to date that has been documented to cause major changes in the composition and structure of reefs (Humann and Deloach 2003). In 2011, Sutherland et al. (2011) were able to definitively identify human waste as the source stressors resulting in white pox disease in elkhorn corals.

While the dominant mode of reproduction for elkhorn and staghorn corals is asexual fragmentation allows rapid recovery from physical disturbances such as storms, this mode of reproduction makes recovery from disease or bleaching episodes (in which entire colonies or even entire stands are killed) very difficult. The large role of asexual reproduction in both species increases the likelihood that genetic diversity in remnant populations may be very low. As broadcast spawners once colonies become rare, the distance between colonies may limit fertilization success and there is substantial evidence to suggest that sexual recruitment of staghorn corals is currently compromised. Reduced colony density in some areas is compounded by low genotypic diversity, indicating that fertilization success and consequently, larval availability, is likely reduced. This can have long-term implications for genetic variability of remaining colonies due to the reduced potential for exchange of genetic material between populations that are spatially further apart (Bruckner 2002).

Both elkhorn and staghorn coral require relatively clear water. The many small polyps and branching morphology of these corals optimizes light capture. This morphology is inefficient for zooplankton capture because zooplankton does not uniformly saturate the water column as light does, so densely arrayed polyps cannot be equally nourished through heterotrophy (Porter 1976). Elkhorn and staghorn corals therefore depend almost entirely upon symbiotic photosynthesizers for nourishment, making them more susceptible to increases in water turbidity and temperature. Different strains of symbiotic zooxanthellae (*Symbiodinium* spp.) can confer different thermal and light tolerances to acroporid (Abrego et al. 2009, Ainsworth and Hoegh-Guldberg 2009, Abrego et al. 2010).

Elkhorn and staghorn corals are also particularly susceptible to damage from sedimentation. Synergistic analyses have found that high temperature increases the risk of colony mortality under a variety of sediment loading conditions, but excessive sediment appears to reduce mortality risk under high light and temperature regimes, possibly by reducing exposure to these stressors (Anthony et al. 2007, Boyett et al. 2007). High sediment with otherwise good light and temperature conditions appears to increase colony mortality (Anthony et al. 2007). High temperature or rapid heating can result in heat shock and alter cellular metabolism within the coral as well as possibly hinder immune response or the ability of zooxanthellae to thrive (Rodriguez-Lanetty et al. 2009, Middlebrook et al. 2010). High sediment with otherwise good light and temperature conditions appears to increase colony mortality.

Designated critical habitat. NMFS published a final rule to designate designated critical habitat for elkhorn and staghorn corals on November 26, 2008. There is a single physical feature that is essential to the conservation of the species: natural consolidated hard substrate or dead coral skeleton that are free from fleshy or turf macroalgae cover and sediment cover. This feature is essential to the conservation of these two species because of the extremely limited recruitment observed and the need for this species to have suitable recruitment habitat.

This designated critical habitat specifies a requirement for substrate of suitable quality and availability, in water depths from the mean high water line to 98 feet (except along some areas of Florida, where 6 foot contour is the shoreward limit), to support successful larval settlement, recruitment, and reattachment of fragments. Four specific areas are designated: the Florida unit, which comprises approximately 3,442 square kilometers of marine habitat (Figure 7); the Puerto

Rico unit, which comprises approximately 3,582 square kilometers of marine habitat; the St. John/St. Thomas unit, which comprises approximately 313 square kilometers of marine habitat; and the St. Croix unit, which comprises approximately 326 square kilometers of marine habitat.

This opinion addresses NNC for the Palm Beach County Intracoastal Waterway, which lies inland from designated critical habitat (Figure 8).

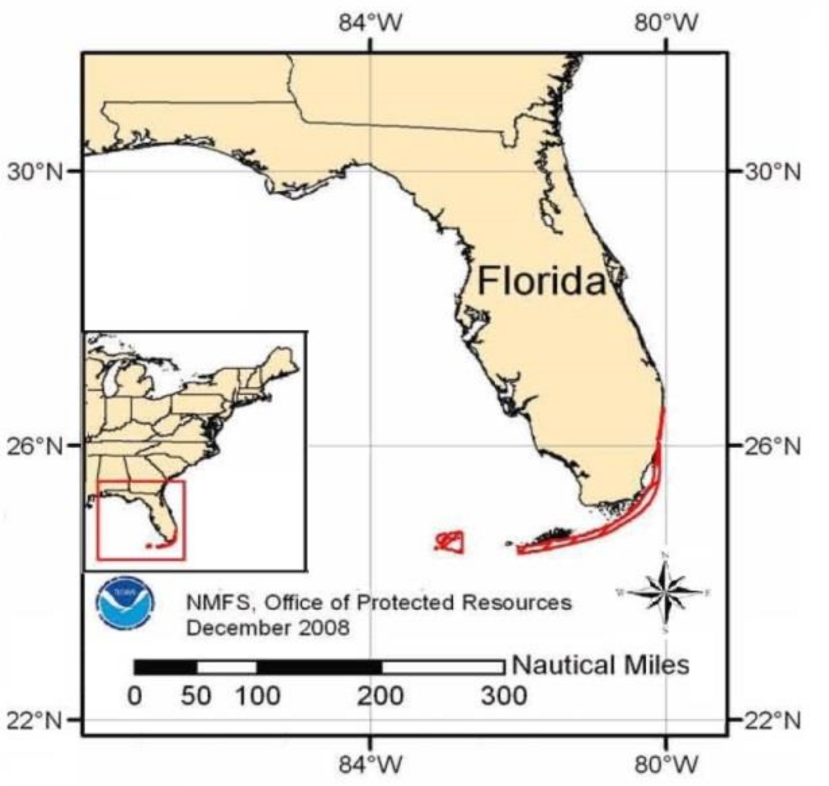

Figure 7. *Acropora* Designated Critical Habitat in Florida.

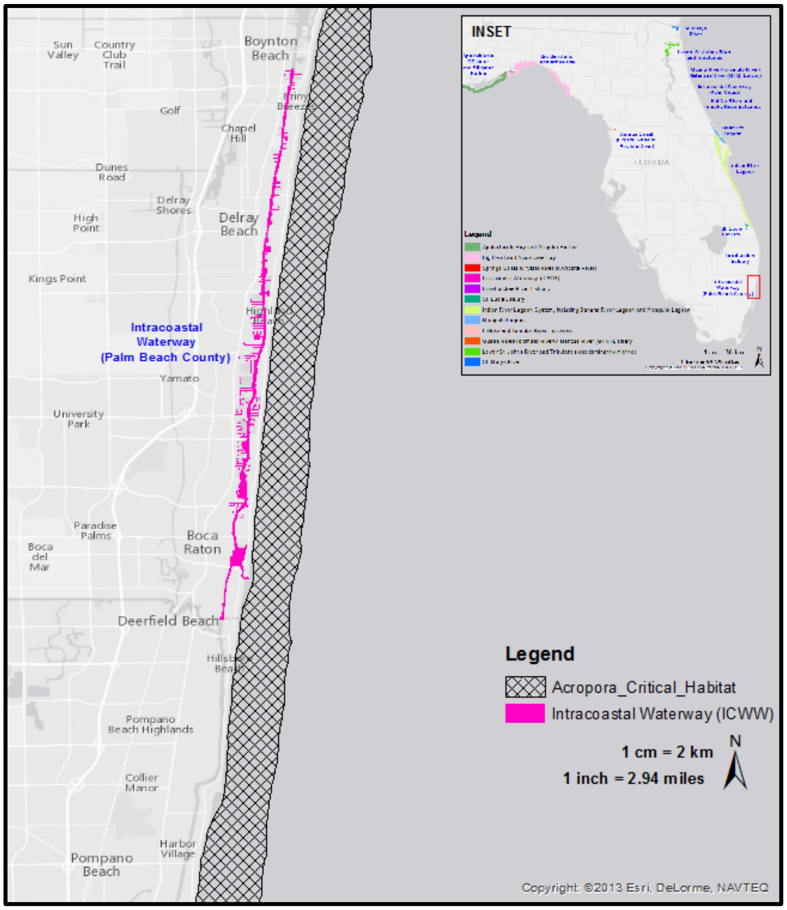

Figure 8. Location of the Palm Beach Intercoastal Waterway Relative to *Acropora* Designated Critical Habitat.

6.4.2 The Orbicella: Lobed Star Coral, Boulder Star Coral, and Mountainous Star Coral

Description. Lobed star coral is distinguished by large, unevenly arrayed polyps that give the colony its characteristic irregular surface. Colony form is variable, and the skeleton is dense with poorly developed annual bands (Weil and Knowlton 1994). Colony diameter can reach up to 5 m with a height of up to 2 m. Common colors are green, grey, and brown (Szmant et al. 1997).

Boulder star coral colonies grow in columns that exhibit rapid and regular upward growth. In contrast to the other species, margins on the sides of columns are typically senescent (Weil and Knowlton 1994). Live colony surfaces usually lack ridges or bumps. Corallites on tops of columns are closely packed, uniformly distributed, and evenly exsert, with maximum diameters of mature corallites typically 2.1–2.6 mm.

Mountainous star coral grows in heads or sheets, the surface of which may be smooth or have keels or bumps. Septa are highly exsert and the skeleton is much less dense than in the other two ESA-listed *Orbicella* species (Weil and Knowlton 1994). Colony diameter can reach up to 10 m with a height of 4–5 m (Szmant et al. 1997). Common colors are grey, green, and brownish (Szmant et al. 1997).

Status. The ESA-listed star corals are found throughout the Caribbean Sea, including the Bahamas and Flower Garden Banks. The range is restricted to the west Atlantic and there is no range fragmentation. There is also a reliable record for boulder star coral in Bermuda and some evidence that lobed star coral may be found in Bermuda as well (Veron 2014).

Star coral species have historically been a dominant species on Caribbean coral reefs, characterizing the so-called “buttress zone” and “annularis zone” in the classical descriptions of Caribbean reefs (Goreau 1959). While declines in the two ESA-listed *Acropora* species began in the early-to-mid-1980s, declines in star coral species were first noticed in the Florida Keys during the mid-1970s and became obvious in the 1990s and 2000s, most often associated with combined disease and bleaching events. It should be noted that, given the dramatically low productivity of the star coral species (low growth and extremely low recruitment), any substantial declines in adult populations would suggest increased extinction risk since their capacity for population recovery is extremely limited.

In Florida, the percent cover data from four fixed sites have shown the star coral species have declined in absolute cover from 5 percent to 2 percent in the Lower Keys between 1998 and 2003 as well as 5– 40 percent colony shrinkage and virtually no recruitment (Smith et al. 2008). Earlier studies from the Florida Keys indicated a 31 percent decline of star coral species absolute cover between 1975 and 1982 (Dustan and Halas 1987) at Carysfort Reef and greater than 75 percent decline (from over 6 percent cover to less than 1 percent) across several sites in Biscayne National Park between the late 1970s and 1998–2000 (Dupont et al. 2008). Taken together, these data imply extreme declines in the Florida Keys (80-95 percent) between the late 1970s and 2003, and it is clear that further dramatic losses occurred in this region during the cold weather event in January 2010. Similar declines have also been documented for relatively remote Caribbean reefs. At Navassa Island National Wildlife Refuge, percent cover of star coral species on randomly sampled patch reefs declined from 26 percent in 2002 to 3 percent in 2009, following disease and bleaching events in this uninhabited oceanic island (Miller and Williams 2007). Additionally, two offshore islands west of Puerto Rico showed reductions in live colony counts of 24 percent and 32 percent between 1998/2000 and 2008 (Bruckner and Hill 2009). Cumulatively, decadal-scale declines across these remote islands in the central Caribbean constitute over 85 percent of the populations. In the U.S. Virgin Islands, data show a decline of star coral species from just over 10 percent cover in 2003 to just over 3 percent cover in 2009 following mass bleaching and disease impacts in 2005 (Miller et al. 2009). This degree of recent decline was preceded by a decline from over 30 percent star coral species cover to about 10 percent between 1988 and 2003 (Edmunds and Elahi 2007). Similarly, percent cover of star corals in a marine protected area in Puerto Rico declined from 49 percent to 8 percent between 1997 and 2009 (Hernandez-Pacheco et al. 2011). Data suggest an 80-90 percent decline in star corals species over the past two decades in the main U.S. Caribbean territories. While Bak and Luckhurst (1980) indicated stability in star coral species cover across depths in Curaçao in the mid-1970s, this region has also manifested star coral declines in recent years. Bruckner and Bruckner (2006) documented an 85 percent increase in the partial star coral colony mortality across three reefs in western Curaçao between 1998 and 2005, approximately twice the level for all other scleractinian species combined.

Star coral species' declines in additional locations can be noted. At Glovers Reef, Belize McClanahan and Muthiga (1998) documented a 38-75 percent decline in relative cover of star coral species across different reef zones between 1975 and 1998, and a further 40 percent decline in relative cover has occurred since then (Huntington et al. 2011). In contrast, star coral populations have shown stable status at sites in Columbia between 1998 and 2003 (Rodriguez-Ramirez et al. 2010), although demographic changes in star coral species at both degraded and less-degraded reefs imply some degree of population decline in this region (Alvarado-Chacón and Acosta 2009).

The boulder star coral, in particular, has a very high estimated extinction risk based on very low productivity (growth and recruitment), documented dramatic declines in abundance, its restriction to the degraded reefs of the wider Caribbean region, and its preferential occurrence in shallow habitats (yielding potentially greater exposure to surface-based threats).

Lobed star coral had a marginally lower estimated extinction risk than the other two *Orbicella* species because of its greater distribution in deep and mesophotic depth habitats, which are not expected to be as vulnerable to some surface-based threats. The overall likelihood that boulder star coral will fall below the critical risk threshold by 2100 was estimated to be in the "likely" risk category with a mean likelihood of 78 percent and a standard error (SE) of 7 percent. The overall likelihood that lobed star coral will fall below the critical risk threshold by 2100 was also estimated to be "likely" (mean likelihood 74 percent, SE of 9 percent), as was the extinction risk for mountainous star coral (mean likelihood of 78 percent and a SE of 7 percent).

Growth and reproduction. Hubbard and Scaturo (1985) reported that star coral growth rates were consistently higher in the clear waters than those at a more turbid and sediment rich site, confirming that light and sediment load are controlling factors of growth rates. All three of the ESA-listed star coral species are hermaphroditic broadcast spawners, with spawning concentrated on nights 6–8 following the new moon in late summer (Levitan et al. 2004). Fertilization success is highly linked to the number of colonies observed spawning at the same time (Levitan et al. 2004). Eggs and larvae are small and post-settlement growth rates are very slow, both of which may contribute to extremely low post-settlement survivorship *(Szmant and Miller 2006)*. There may be a depth-related fecundity cost arising from morphological differences in polyp spacing (Villinski 2003), suggesting the spatial distribution of colonies may influence population fecundity on a reef. Studies from throughout the Caribbean report recruitment to be negligible to absent (Bak and Engel 1979, Rogers et al. 1984). Despite their generally massive form, at least the lobate star coral form is capable of some degree of fragmentation/fission and clonal reproduction.

Habitat. Star coral occur in most reef environments (Veron 2000, Carpenter et al. 2008). *Orbicella* spp. are a common, often dominant component of Caribbean mesophotic reefs (Smith et al. 2010), suggesting the potential for deep refugia. Lobed star coral tend to have a deeper distribution than the other two listed species of *Orbicella* (Szmant et al. 1997), occurring in water depths ranging from 5 m to 50 m (Weil and Knowlton 1994, Carpenter et al. 2008, Bongaerts et al. 2010), while boulder star coral occurs in waters 0.5-20 m deep (Szmant et al. 1997) and mountainous star coral typically occurs between 10-20 m in fore-reef environments, although it may be found in 0.5-40 m (Weil and Knowlton 1994, Carpenter et al. 2008).

Threats. Both Bruckner and Bruckner (2006) and Miller et al. (2009) demonstrated profound population declines for star coral species from disease impacts, both with and without prior bleaching. Both white plague and yellow-band diseases can invoke this type of population level decline. Disease outbreaks can persist for years in a population—star coral colonies suffering from yellow-band in Puerto Rico still manifested similar disease signs four years later, with a mean tissue loss of 60 percent (Bruckner and Bruckner 2006).

Star corals species do not suffer from catastrophic outbreaks of predators. While star corals can host large populations of corallivorous snails, they rarely display large feeding scars that are apparent on other coral prey, possibly related to differences in tissue characteristics or nutritional value (Baums et al. 2003). However, low-level predation can have interactive effects with other stressors. For example, predation by butterflyfish can serve as a vector to facilitate infection of mountainous star coral with black-band disease (Aeby and Santavy 2006). Parrotfishes are also known to preferentially target star corals species in so-called “spot-biting” which can leave dramatic signs in some local areas (Bruckner et al. 2000, Rotjan and Lewis 2006), and chronic parrotfish biting can impede colony recovery from bleaching (Rotjan et al. 2006). Although it is not predation per se, star coral colonies have often been infested by other pest organisms. Bioeroding sponges (Ward and Risk 1977) and territorial damselfishes can cause tissue loss and skeletal damage. Damselfish infestation of star coral species appears to have increased in areas where their preferred, branching coral habitat has declined because of loss of other species (Precht et al. 2010).

The only study conducted regarding the impact of acidification on this genus is a field study (Helmle et al. 2011) that did not find any change in field-sampled colonies of mountainous star coral calcification in the Florida Keys through 1996. Recent work in the Mesoamerican reef system indicated that mountainous star corals had reduced thermal tolerances in locations and over time (Carilli et al. 2010) with increasing human populations, implying increasing local threats (Carilli et al. 2009).

Published reports of individual bleaching surveys have consistently indicated that star coral species are highly-to-moderately susceptible to bleaching (Oxenford et al. 2008, Brandt 2009, Bruckner and Hill 2009, Wagner et al. 2010). Star corals can contain multiple varieties of zooxanthellae, depending on depth and other environmental conditions (Rodriguez-Roman et al. 2006, Thornhill et al. 2006). Bleaching has been shown to prevent gametogenesis in star coral colonies in the following reproductive season after recovering normal pigmentation (Szmant and Gassman 1990, Mendes and Woodley 2002) and leave permanent records in coral growth records (Leder et al. 1991, Mendes and Woodley 2002).

Particularly well documented mortalities in star coral species following severe mass bleaching in 2005 highlight the immense impact that thermal stress events and their aftermath can have (Miller et al. 2009). Hernandez-Pacheco et al. (2011) showed that demographic transitions (vital rates) for star coral species were substantially altered by the 2005 mass thermal bleaching event. Size-based transition matrix models based on these measured vital rates showed that population growth rates were stable (λ not significantly different from 1) in the pre-bleaching period (2001–2005) but declined to $\lambda = 0.806$ one year after and to 0.747 two years after the bleaching event. Although population growth rate returned to $\lambda = 1$ the following year, simulation modeling of different bleaching probabilities predicted extinction of a population with these dynamics within 100 years at a bleaching event once every 5-10 years (Hernandez-Pacheco et al. 2011). Cervino et al. (2004) also showed that higher temperatures (20–31° C) resulted in faster rates of tissue loss and higher mortality in yellow-band affected star coral species.

Tomascik and Logan (1990) found a general pattern of decreasing growth rates over the past 30 years at seven fringing reefs along the west coast of Barbados and contributed this decrease to the deterioration of water quality. Average growth rate of star coral species increased with improving water quality conditions on fringing reefs in Barbados. Torres and Morelock (2002) noted a similar decline in star coral species growth at sediment-impacted reefs in Puerto Rico. Density and calcification rate increased from high to low turbidity and sediment load, while extension rate followed an inverse trend (Carricart-Ganivet and Merino 2001). Eakin et al. (2010) demonstrated declines in star coral species linear extension during periods of construction in Aruba. Downs et al. (2005) suggested that localized toxicant exposure may account for a localized mortality event of star coral species in Biscayne National Park. Mountainous star coral induces the toxicant-metabolizing enzyme cytochrome p450 and antioxidant enzymes under acute exposure to benzo(a)pyrene (Ramos and Garcia 2007), but effects of chronic long-term exposure are not known. Star coral species' skeletons are among those that incorporate toxic heavy metals, making them useful in documenting long-term contamination of reef sites (Medina-Elizalde et al. 2002, Runnalls and Coleman 2003). Nutrient-related runoff has also been deleterious to star coral species. Elevated nitrogen reduced respiration and calcification in star coral *and* stimulated zooxanthellae populations (Marubini and Davies 1996). Fecal coliform microorganisms were among the bacterial communities associated with *Orbicella* in the Florida Keys (Lipp et al. 2002), suggesting potential sewage impacts to the corals. Elevated nutrients increased the rate of tissue loss in star coral species affected by yellow-band disease (Bruno et al. 2003). Chronic nutrient elevation can produce bleaching and partial mortality in star coral species, whereas anthropogenic dissolved organic carbon kills corals directly (Kuntz et al. 2005).

Designated critical habitat. Designated critical habitat has not been proposed for any of the *Orbicella* spp.

6.4.3 Pillar coral

Description. Pillar coral colonies have encrusting bases on which cylindrical columns are developed that may reach 2 m in height. Valleys are meandroid. Tentacles remain extended during the day giving columns a furry appearance. Colonies are generally grey-brown in color (Veron 2000)

Status. Pillar coral is reported to be uncommon (Veron 2000) with isolated colonies scattered across a range of habitat types. Overall colony density throughout south Florida was estimated to be about 0.6 colonies per 10 m^2 (Wagner et al. 2010), while it was estimated to be 172 ± 177 ind/m2, mean density 172 per square kilometers in the Columbian Caribbean (Acosta and Acevedo 2006). Pillar coral is restricted to the west Atlantic where it is present throughout the greater Caribbean but is one of the Caribbean genera absent from the southwest Gulf of Mexico (Tunnell 1988). Pillar coral occurs in south Florida and the U.S. Caribbean but appears to be absent from the Flower Garden Banks.

Growth and reproduction. Pillar corals have separate male and female colonies that release gametes that float and create a sheen on the water (Szmant 1986). This “gonochoric” spawning coupled with persistently low population densities results in low probability for successful fertilization and, therefore, larval supply. No juvenile pillar coral were observed in surveys of 566 sites in the Florida Keys during 1999–2009 (Miller et al. 2011), in larval settlement studies in the U.S. Virgin Islands in the early 1980s (Rogers et al. 1984), or in juvenile surveys in the mid-1970s in the Netherlands Antilles (Bak and Engel 1979). Propagation of pillar coral by fragmentation following storms or other physical disturbances is the likely source of unexpected aggregations of colonies (Hudson and Goodwin 1997). Annual growth rates of 12–20 mm per year in linear extension have been reported (Hudson and Goodwin 1997), but up to 80 mm annually have been reported (Hughes and Connell 1987, Acosta and Acevedo 2006). Partial mortality rates have been size-specific but generally low (Acosta and Acevedo 2006). Feeding clearance rates are low (Lewis 1976), but pillar coral has a relatively high photosynthetic rate and stable isotope values suggest it receives substantial amounts of photosynthetic products translocated from its zooxanthellae (Muscatine and Kaplan 1994).

Habitat. Pillar coral inhabits most reef environments (Veron 2000), but in the Florida Keys it appears to be absent in nearshore hard bottoms, nearshore patch reefs, and backreef environments and more common on forereef spur-and-groove habitats (Miller et al. 2011). Pillar coral has been reported in water depths ranging from 2-25 m (Carpenter et al. 2008).

Threats. There are conflicting characterizations of bleaching susceptibility of pillar coral in the literature. The species was bleaching-resistant during the 1983 mass bleaching event in Florida (Jaap 1985). Characterizations of the 2005 mass bleaching event in southern Florida and in the U.S. Virgin Islands noted that no bleached pillar coral colonies were observed (Clark et al. 2009, Wagner et al. 2010). In contrast, Oxenford et al. (2008) report that 100 percent of the 15 colonies they observed in Barbados during the 2005 mass bleaching event were bleached. Pillar coral is sensitive to cold shock in the Caribbean (Muscatine et al. 1991).

Black-band disease can affect pillar coral colonies (Ward et al. 2006), but white plague causes more extensive impacts, which can cause rapid tissue loss (Miller et al. 2006). The large colony size suggests that individual colonies are less likely to suffer complete mortality from a given disease exposure, but low colony density in this species suggests that even small degrees of mortality increase extinction risk.

The corallivorous fireworm has been observed on diseased colonies of pillar coral (Miller et al. 2006), but, generally, predation is not observed to cause noticeable mortality.

Bak and Elgershuizen (1976) found that the rate of sand removal from pillar coral tissues in laboratory conditions was intermediate among 19 Caribbean coral species tested. Along a eutrophication gradient in Barbados, pillar coral was found at only a single site—one of those farthest removed from pollution (Tomascik and Sander 1987a).

Given the apparent naturally rare status of this species, some undescribed adaptations to low population density may exist in this species (particularly with regard to overcoming fertilization limitation between spawned gametes from gonochoric parent colonies that are at great distance from one another (Brainard et al. 2011).

The overall likelihood that pillar coral will fall below the critical risk threshold by 2100 was estimated to be in the “likely” risk category with a mean likelihood of 74 percent and a SE of 6.6 percent (Brainard et al. 2011).

Designated critical habitat. Designated critical habitat has not been proposed for pillar coral.

6.4.4 Rough Cactus Coral

Description of the species. Rough cactus coral consists of encrusting laminar plates. Colonies are thin, weakly attached plates with interconnecting, slightly sinuous narrow valleys. Corallite centers are usually in single rows. Columellae are rudimentary or absent. Colonies are most commonly greys and browns, with valleys and walls of contrasting colors. Maximum colony size is 50 cm (Veron 2000).

Status. Rough cactus coral occurs along the southern tip of Florida and the Florida Keys. Disease has resulted in population declines over the past several decades in the Florida Keys. Rough cactus coral is uncommon (Veron 2000), constituting < 0.1 percent of coral colonies and occurs at densities < 0.8 colonies per 10 m^2 in Florida (Wagner et al. 2010) and at 0.8 colonies per 100 m transect in Puerto Rico sites (http://www.agrra.org). Monitoring data since 2000 from Florida, Puerto Rico, and St. Croix show rough cactus coral cover to be consistently less than 1 percent, with occasional observations up to 2 percent (available online at http://www8.nos.noaa.gov/biogeo_public/query_habitat.aspx). Dustan (1977) suggests that rough cactus coral was much more abundant in the upper Florida Keys in the early to mid-1970s than currently. The overall likelihood that rough cactus coral will fall below the critical risk threshold by 2100 was estimated to be in the “likely” risk category with a mean likelihood of 70 percent and a SE of 8 percent (Brainard et al. 2011).

Growth and reproduction. Rough cactus coral is hermaphroditic and a brooder. Polyps produce 96 eggs per cycle on average (Szmant 1986). Their larvae contain zooxanthellae that can supplement maternal provisioning with energy sources provided by their photosynthesis (Baird et al. 2009). Colony size at first reproduction is > 100 cm^2 (Szmant 1986). Recruitment appears to be very low (Dustan 1977).

Habitat. Rough cactus coral has been reported to occur in shallow reef environments (Veron 2000) ranging from 5-30 m (Carpenter et al. 2008).

Threats. No bleached rough cactus coral colonies were observed during the 2005 mass coral bleaching event in Florida (Wagner et al. 2010) or Barbados (Oxenford et al. 2008), although the number of colonies was small in Barbados.

Rough cactus coral are susceptible to acute and subacute white plague. Dustan (1977) reported dramatic impacts from this disease to the population in the upper Florida Keys in the mid-1970s. He also reported that the rate of disease progression was positively correlated with water temperature and measured rates of disease progression up to 3 mm daily. Rough cactus coral were absent at fringing reef sites impaired by sewage pollution (Tomascik and Sander 1987a).

Designated critical habitat. Designated critical habitat has not been proposed for rough cactus coral.

6.5 Johnson's Seagrass

Description. Johnson’s seagrass is a relatively small seagrass species with paired linearly shaped spatulate leaves with smooth margins. The leaves are 0.2-1.0 inches (0.5-2.5 cm) long and growing from a creeping rhizome with petioles, sessile (that is, attached to their bases) female flowers, and long-necked fruits. The male flowers are unknown.

Status and trends. On September 14, 1998, Johnson's seagrass was listed as threatened under the ESA. Historical abundance estimates of Johnson’s seagrass are not available due to the species having only recently been differentiated. Limited data indicate no large distributional gaps or changes in abundance over much of Johnson’s seagrass distribution from 1994 to 1999. However, recent increases in reported occurrence could be an artifact of recent increases in search efforts.

The species has only relatively recently been identified as a distinct species and therefore no historical distribution information is available (Eiseman and McMillan 1980). Current distribution includes lagoons along approximately 201 kilometers of southeastern Florida between Sebastian Inlet and north Biscayne Bay, which means that Johnson’s seagrass has the most limited geographic distribution of any seagrass in the world (Kenworthy 1997). However, northern range extensions (likely temporary) have recently been observed (Virnstein and Hall 2009). The largest known groups of patches are located near Sebastian Inlet and Lake Worth, Florida.

Habitat. Patches of Johnson’s seagrass have been observed to grow from the intertidal zone down to 3.3 feet water depth and in waters with variable temperatures and salinities (15 to 43 parts per thousand) and temperatures (Dawes et al. 1989, Kenworthy 1993, Virnstein et al. 1997, Kahn and Durako 2009). Patches near freshwater discharges have been observed (Gallegos and Kenworthy 1996), although Torquemada et al. (2005b) noted that highly hypo- or hypersaline conditions can negatively impact growth. Intertidal patches may be completely exposed at low tides, suggesting tolerance to desiccation and wide temperature ranges (Kahn and Durako 2009).

Growth and reproduction. Only female flowers have been observed; no fruit or seeds have been found to date (Eiseman and McMillan 1980, Heidelbaugh et al. 2000). However, there is no evidence of male flowers, meaning Johnson's seagrass probably reproduces by cloning or asexual branching and fragmentation (Jewitt-Smith et al. 1997, Hammerstom and Kenworthy 2003). Consequently, genetic diversity is low (Freshwater and York 1999), putting Johnson's seagrass at a potential genetic disadvantage compared to other seagrasses.

Clonal reproduction occurs when plants form new leaf-pair, root and rhizome segments that arise from terminal buds (Posluszny and Tomlinson 1990). As clones expand, high density "patches" are formed ranging from three to 66 feet^2 in size (Kenworthy 1997, Virnstein et al. 1997, Kenworthy 2000, 2003, Virnstein and Morris 2007). Patches can expand rapidly at about nine feet^2 per month (Kenworthy 2003) leading to coalescence with adjacent patches and large meadows of up to 0.12 square kilometers (Kenworthy 1997).

Based on 14 years of data collected in the Indian River Lagoon, Virnstein et al. (2009) described the species as "sparse, but persistent... with accompanying high variability so that regional stability over time is maintained by local unpredictability" suggesting that the long term survival of the species is based on spatially and temporally asynchronous pulsating patches of abundance. The species appears to avoid direct competition with taller canopy forming seagrasses. Johnson's seagrass appears to be physiologically adapted to exploit unstable environments and unvegetated patches, with minimal resources allocated to the holding of space (Dean and Durako 2007). Fragments or entire plants can be uprooted and drift extensively, providing a mechanism for dispersal and colonization of new areas (Hall et al. 2006). Johnson's seagrass frequently undergo whole patch mortality followed by recolonization (Virnstein et al. 1997, Heidelbaugh et al. 2000, Greening and Holland 2003, Kenworthy 2003, Virnstein and Morris 2007). Although successful in unstable areas, Johnson's seagrass may be out-competed by more stable-selected plants in areas not subject to regular disturbance (Durako 2003). Due to this species' physiology, low capacity for storage, and shallow root system, growth over large unsuitable patches may be unlikely, and its ability to recover from widespread habitat loss may be limited.

Threats. Storms pose the greatest natural threat to Johnson's seagrass. Storms can easily uproot or rip apart individuals and scatter them widely. Although this can serve to disperse individuals into new habitats, it can also catastrophically eliminate established meadows. Subsequent siltation following high turbidity events such as algal blooms can also bury individuals or parts of plants. Due to its delicate morphology, small range, lack of genetic diversity and a physiology ill equipped to hold space and compete with other seagrasses, Johnson's seagrass is vulnerable to prolonged widespread human-induced disturbance and habitat loss and its potential for recovery may be limited. The growth of boating in Florida and development of coastal areas has resulted in trampling, propeller scarring, dredging, filling, shading, and altered water quality that has degraded these areas compared to historical conditions. The species is under threat from high development pressure and subsequent habitat degradation throughout its range. Johnson’s seagrass and its habitat are threatened by several specific natural and anthropogenic factors, including (1) dredging and filling, (2) construction and shading from in- and overwater structures, (3) prop scarring and anchor mooring, (4) trampling, (5) altered water quality (such as stormwater runoff and turbidity), (6) siltation, and (7) climate change (M et al. 2009).

Designated critical habitat. Designated critical habitat for Johnson’s seagrass was designated on April 5, 2000. The designated critical habitat occurs entirely within Florida and includes (1) locations with populations that have persisted for 10 years; (2) locations with persistent flowering populations; (3) locations at the northern and southern range limits of the species; (4) locations with unique genetic diversity; and (5) locations with a documented high abundance of Johnson’s seagrass compared to other areas in the species’ range. These are critical to the conservation of the species because they protect persistently reproductive and genetically diverse populations, allow for protective buffers along the distribution limits (i.e., edges of survival), and protect regions of high density that without further knowledge of species biology, appear to serve the needs of Johnson’s seagrass. Ten regions of sheltered bay and inlet waters are designated, including north and south of Sebastian Inlet, near Fort Pierce Inlet, north of St. Lucie Inlet, a portion of Hobe Sound, the southern side of Jupiter Inlet, Lake Worth Lagoon (north of Bingham Island and Boynton Inlet), waters of Lake Wyman, and wide areas of northern Biscayne Bay. These regions occupy approximately 91.39 square kilometers (Figure 9). The nature of Johnson’s seagrass designated critical habitat makes it variable and prone to change.

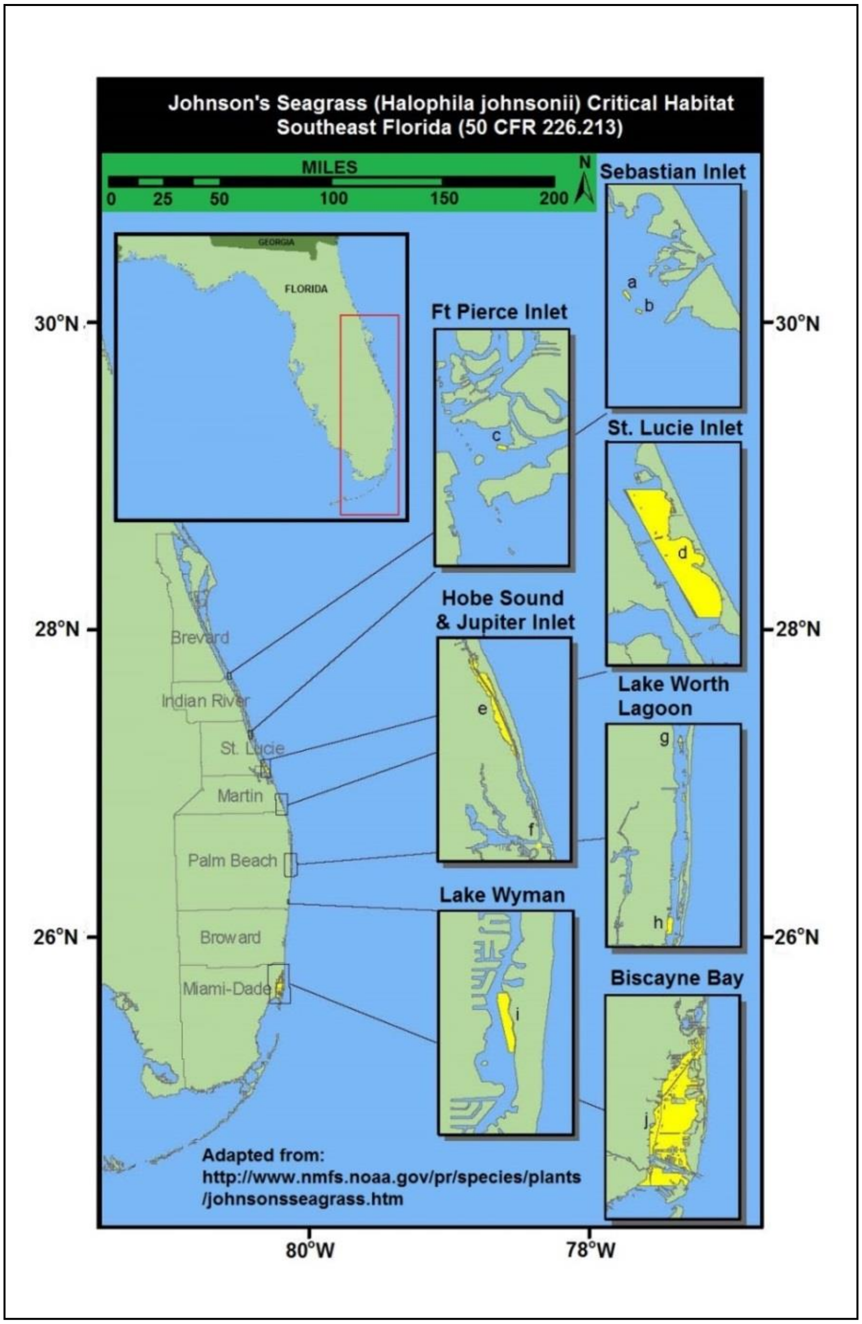

Figure 9. Johnson's Seagrass Designated Critical Habitat. a) North of Sebastian Inlet Channel, b) South of Sebastian Inlet Channel, c) Ft. Pierce Inlet, d) North of St. Lucie Inlet, e) Hobe Sound, f) South Side of Jupiter Inlet, g) A Portion of Lake Worth Lagoon North of Bingham Island, h) A Portion of Lake Worth Lagoon, Located Just North of Boynton Inlet, i) a Portion of Northeast Lake Wyman, Boca Raton, j) a Portion of Northern Biscayne Bay.

7 ENVIRONMENTAL BASELINE

The *Environmental Baseline* includes the past and present impacts of all Federal, state, or private actions and other human activities in the action area, the anticipated impacts of all proposed Federal projects in the action area that have already undergone formal or early section 7 consultation, and the impact of state or private actions which are contemporaneous with the consultation in process (50 CFR 402.02). The key purpose of the *Environmental Baseline* is to describe the condition of the ESA-listed species and designated critical habitat within the action area and the consequences of that condition without the action.

Baseline conditions nationwide are reflected within Florida. Flather et al. (1998) identified habitat loss and alien species as the two most widespread threats to endangered species, affecting more than 95 percent and 35 percent of listed species, respectively. For example, the net effect of human-altered hydrology creates conditions that increase stormwater runoff; transporting land based pollutants into surface waters and reduce the filtration of stormwater runoff through wetlands prior to reaching surface waters. As a result, altered hydrology has reduced the spatial extent and quality of available habitat and the connectivity among rivers and streams that is necessary for anadromous species to complete their migratory lifecycles.

Increases in polluted runoff have been linked to a loss of aquatic species diversity and abundance, including many important commercial and recreational fish species. Non-point source pollution has also contributed to coral reef degradation, fish kills, seagrass bed declines, and algal blooms, including blooms of toxic algae. In addition, many shellfish bed and swimming beach closures can be attributed to polluted runoff. As discussed in EPA’s latest National Coastal Condition Report, non-point sources have been identified as one of the stressors contributing to coastal water pollution (USEPA 2012a).

With its mean elevation above sea level of 30 meters and porous limestone aquifers, Florida is uniquely vulnerable to sea level rise associated with climate change. Expansion of inland tidal marshes replacing lowland coastal forests over the last 120 years was demonstrated along the Big Bend of Florida (Raabe and Stumpf 2016). Temperature records between 1878 and 2012 for Florida Keys coral reef habitats indicate an increase of 0.8^{o}C in the last century (Kuffner et al. 2015).

The Intergovernmental Panel on Climate Change estimated that average global land and sea surface temperature has increased by 0.85°C (± 0.2) since the late 1800s, with most of the change occurring since the mid-1900s (IPCC 2013). This temperature increase is greater than what would be expected given the range of natural climatic variability recorded over the past 1,000 years (Crowley and Berner 2001). All species discussed in this opinion are presently, or are likely to be, affected by the direct and indirect effects of global climatic change. Global climate change stressors, including consequent changes in land use, are major drivers of ecosystem alterations (USEPA 2008). Climate change is projected to have substantial direct effects on individuals, populations, species, and the community structure and function of marine, coastal, and terrestrial ecosystems in the foreseeable future (McCarty 2001, IPCC 2002, Parry et al. 2007, IPCC 2013). A northward shift in loggerhead nest placement was reported for Melborne Beach, Florida, the largest U.S. rookery for this species (Reece et al. 2013). Climate change is most likely to have its most pronounced effects on species whose populations are already in tenuous positions (Williams et al. 2008). Increasing atmospheric temperatures have already contributed to changes in the quality of freshwater, coastal, and marine ecosystems and have contributed to the decline of populations of endangered and threatened species (Mantua et al. 1997, Karl et al. 2009, Littell et al. 2009).

Increasing surface water temperatures can cause the latitudinal distribution of freshwater and marine fish species to change: as water temperatures rise, cold and warm water species will spread northward (Hiddink and ter Hofstede 2008, Britton et al. 2010). Climate-mediated changes in the global distribution and abundance of marine species are expected to reduce the productivity of the oceans by affecting keystone prey species in marine ecosystems such as phytoplankton, krill, and cephalopods. For example, climate change may reduce recruitment in krill by degrading the quality of areas used for reproduction (Walther et al. 2002). Aquatic nuisance species invasions are also likely to change over time, as oceans warm and ecosystems become less resilient to disturbances (USEPA 2008). Invasive species that are better adapted to warmer water temperatures could outcompete native species that are physiologically geared towards lower water temperatures; such a situation currently occurs along central and northern California (Lockwood and Somero 2011). Warmer water stimulates biological processes, which can lead to environmental hypoxia. Oxygen depletion in aquatic ecosystems can result in anaerobic metabolism increasing, thus leading to an increase in metals and other pollutants being released into the water column (Staudinger et al. 2012).

Ocean acidification, as a result of increased atmospheric carbon dioxide, can interfere with numerous biological processes in corals including fertilization, larval development, settlement success, and secretion of skeletons (Albright et al. 2010). In addition to global warming, acidification poses another significant threat to oceans because many major biological functions respond negatively to increased acidity of seawater. Photosynthesis, respiration rates, growth rates, calcification rates, reproduction, and recruitment may be negatively impacted with increased ocean acidity (Royal Society of London 2005). Kroeker et al. (2010) review of 139 studies quantifying ocean acidification effects determined that the effects were variable depending on species, but effects were generally negative, with calcification being one of the most sensitive processes.

Aquatic species, especially marine species, already experience stress related to the impacts of rising temperature. Corals, in particular, demonstrate extreme sensitivity to even small temperature increases. When sea temperatures increase beyond a coral’s limit, the coral “bleaches” by expelling the symbiotic organisms that not only give coral its color, but provide food for the coral through their photosynthetic capabilities. According to (Hoegh-Guldberg 2010), bleaching events have steadily increased in frequency since the 1980s.

7.1 Baseline Condition of Florida's Aquatic Resources

The baseline condition of Florida's aquatic resources is described in detail in the 2014 Integrated Water Quality Assessment for Florida (FDEP 2014a). The following paragraphs are derived from that document. There are 88,250 km of streams and rivers, 127,239 kilometers of ditches and canals, and 45,838 square kilometers of freshwater and tidal wetlands in Florida (Figure 10). Florida's coastline ranks second in length only to Alaska. Florida's low relief, coupled with its geologic history, has created unique hydrogeologic features making groundwater quality particularly critical to surface water quality.

7.1.1 Human Alterations of Surface Waters

Major dams have been built on the Apalachicola, Ocklawaha, Ochlockonee, Hillsborough, Withlacoochee (Citrus County) Rivers, and, Ocklawaha River and channelizing the Kissimmee River. The hydrology of the southern third of Florida's peninsula has been significantly altered, and few naturally flowing streams and rivers remain. Most fresh waterbodies in south Florida are canals. Several efforts are under way to reverse some of the alterations, thus restoring natural flows and function to waterbodies. Significant work on the Kissimmee River since the 1990s has successfully restored flow in portions of the historical river channel, leading to improved habitat, fisheries, and water quality.

In the past, many wetlands were drained for agriculture, logging, and urban development, and numerous rivers were channelized for navigation. The modifications were most intense in south Florida, where, beginning in the 1920s, canals, and levees were built to control flooding and to drain wetlands. These modifications resulted in the loss of much of the original Everglades wetlands from Lake Okeechobee south. The Everglades restoration under way is intended to improve water quality. There are preliminary successes; however, restoration is a long-term effort involving many agencies working to revitalize the heavily altered system. The logging and agricultural activities that were once occurred along the St. Marys River are no longer pursued and the area and the St. Marys River has rejuvenated, The river is currently a popular area for recreation and sightseeing. Intense development along the St. Johns River contributed pollutants through stormwater, wastewater discharges, and agricultural runoff.

Currently the Port of Miami is being dredged to accommodate the newest generation of freighters. Among sediment impacts assessed, the most severe is for a sedimentation assessment site located 200 m north of the dredged channel. This assessment characterized 81 percent of the points surveyed as 'sediment over hard bottom' compared to 1 percent at the corresponding reference site.

from http://www.thepalmbeachtimes.com/TravelNavigator/FloridaMileage.php

Figure 10. Map[5] of Major Surface Waters in Florida.

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[5] Adapted from http://www.thepalmbeachtimes.com/TravelNavigator/FloridaMileage.php. Added north arrow, labeled Ten Thousand Islands and Indian River, recolored and relocated scale bar for legibility.

7.1.2 Pollutants

Arsenic has recently arisen as the pollutant of concern. The Tampa Bay Tributaries, Withlacoochee, Sarasota–Peace–Myakka, and Ocklawaha Basins have had the highest number of water systems reporting samples with elevated arsenic. The basins with the highest number of wells with exceedances for the two-year period associated with the Tampa Bay Tributaries, Suwannee, Withlacoochee, and Springs Coast Basins. Arsenic in ground water may be naturally occurring, of anthropogenic origin due to human-induced geochemical changes, or a true contaminant released as a result of human activities. The prevalence of elevated arsenic detections in the southwest Florida basins and the Suwannee Basin may be due to the chemical makeup of the aquifer in these areas.

In addition to this natural source, potential anthropogenic sources include arsenic-based pesticides applied to cotton fields; citrus groves; road, railroad, and power line rights-of way; golf courses; and cattle-dipping vats, which were in use in Florida until 1961 (McKinnon et al. 2011). In recent years, the use of arsenical pesticides has significantly decreased, and as of 2013 its use is restricted only to monosodium methanearsonate on cotton fields, golf courses, sod farms, and highway rights-of-way (78 CFR 59). However, residues from past use, when bound to soil particles, do not readily dissipate. Higher numbers of reported exceedances may be considered an artifact of the change in the EPA arsenic standard for ground water, which was reduced from 50 to 10 μg/L in 2001, and was fully implemented in 2006.

Activities such as mining, well drilling, stormwater discharge into drainage wells, aquifer storage and recovery projects (Arthur et al. 2002, Price and Pichler 2006), and over pumping can potentially release previously stable arsenic into ground water. In addition, drought can lower the water table, allowing oxygen to permeate the aquifer matrix and cause the release of arsenic compounds from limestone.

Ground water contamination by nitrate remains an ongoing problem and a challenge to water resource managers in Florida. One effort to reduce fertilizer leaching into wells is the implementation of agricultural best management practices by farmers. Another aspect that may be reducing contamination is the transition from agricultural to residential land uses, resulting in less fertilizer use in some agricultural areas. In addition, in some of these transitioning areas, public water supplies have become available to homeowners who were previously on individual wells. The combination of reduced sources and reduced number of wells requiring monitoring may be partially responsible for the decrease in the number of wells found to be contaminated in recent years.

In aquatic environments, sediments provide essential habitat but, at the same time, may be a source of contamination and recycled nutrients. Sediment contaminants, such as trace metals, organic pesticides, and excess nutrients, accumulate over time from upland discharges, the decomposition of organic material, and atmospheric deposition. Periodic water quality monitoring cannot fully evaluate aquatic ecosystems, as it is not usually designed to assess the cumulative impact of sediment contaminants. Knowledge of a site's sediment quality is important for environmental managers in evaluating future restoration and dredging projects. Unlike many water column constituents, the Florida has no criteria for sediment and no statutory authority to establish criteria. Therefore, it is important to use scientifically defensible thresholds to estimate the condition of sediments and determine the ecological significance of these thresholds.

7.1.3 Fisheries Bycatch

Bycatch occurs when fishing operations discard fish or interact with marine mammals, sea turtles, protected fish species, corals, sponges, or seabirds. Bycatch is the primary reason for the decline and, ultimately, the listing of smalltooth sawfish as endangered in 2003 (NMFS 2009) The long, toothed rostrum of the smalltooth sawfish causes this species to be particularly vulnerable to entanglement in fishing nets. Historical reports of smalltooth sawfish caught in otter trawls, trammel nets, and seine nets were relatively common in Florida and other areas in the Gulf of Mexico (NMFS 2009). Bigelow and Schroeder (1953), who described smalltooth sawfish as “plentiful in Florida waters,” noted they were of “considerable concern to fishermen as nuisances because of the damage they do to drift- and turtle nets, to seines, and to shrimp trawls in which they often become entangled and because of the difficulty of disentangling them without being injured by their saws.” Smalltooth sawfish bycatch in shrimp trawl operations declined rapidly in the second half of the 20th century due to population decline. In Louisiana shrimp trawl landings, which were reported as high as 34,900 pounds in 1949, dropped to zero landings recorded after 1978 (Simpfendorfer 2002). In Florida, smalltooth sawfish have only occasionally been recorded in shrimp trawl landing since the 1990s (NMFS 2009). Smalltooth sawfish are also caught incidentally in shark drift gillnet and shark bottom longline fisheries, although interactions with these fisheries are considered relatively rare. A 2003 Highly Migratory Species Opinion estimated one incidental capture of a sawfish every five years in the shark gillnet fishery (NMFS 2003). An estimated 61 smalltooth sawfish were captured in the Atlantic and Gulf of Mexico shark bottom longline fishery from 2005-2006 (NMFS 2011). Smalltooth sawfish are also caught incidentally by recreational anglers, particularly within the Everglades National Park. However, such interactions are considered very rare and the impacts to the species associated with post-release mortality are probably small (NMFS 2009).

Sturgeon bycatch estimates based on NMFS' ocean observer data are considered to be underestimated since bycatch is underreported in state waters and no observer coverage exists in South Atlantic (North Carolina-Florida) U.S. Federal waters (ASMFC Technical Committee 2006). Commercial fishery bycatch data for other waters indicate that bycatch is a significant threat to the viability of listed sturgeon species and populations (ASSRT 2007, SSSRT 2010). Although directed harvest of Atlantic, shortnose, and Gulf sturgeon is prohibited, these species are still incidentally caught in several commercial fisheries operating throughout their ranges. Shortnose sturgeon are primarily captured in gillnets but have also been caught in pound nets, fyke/hoop nets, catfish traps, shrimp trawls and hook and line fisheries (recreational angling). Bycatch of shortnose sturgeon from shad gillnet fisheries can result in a significant source of mortality (SSSRT 2010). In one study from South Carolina, out of 51 shortnose sturgeon captured, 16 percent resulted in bycatch mortality and another 20 percent were visibly injured (Collins et al. 1996). Bycatch could also have a substantial impact on the status of Atlantic sturgeon, especially in rivers or estuaries that do not currently support a large subpopulation (< 300 spawning adults per year). Estimated bycatch mortality rates for Atlantic sturgeon range from 0 percent-51 percent depending on gear and other conditions, with greatest mortality occurring in sink gill nets (ASSRT 2007). Inland American shad gill net fisheries in two southern locations (Winyah Bay and Altamaha River) were estimated to capture 530 Atlantic sturgeon, of which 58 likely resulted in mortality. Atlantic sturgeon mortality associated with bycatch has been estimated as high as 1,400 deaths per year from 1989–2000 in the ocean fisheries ranging from North Carolina to Maine (Stein et al. 2004a).

Commercial fishing operations have been identified as the most significant source of injury and mortality of juvenile, subadult, and adult sea turtles and a major threat that has contributed to ESA listings of several sea turtle species. Bycatch of sea turtles in shrimp trawl fisheries conducted off the southeast United States (from North Carolina to the Atlantic coast of Florida) and U.S. Gulf of Mexico (from the Gulf coast of Florida to Texas) can result in significant demographic effects on sea turtle populations. Participants in these fisheries have been required to use turtle exclusion devices since 1987. Turtle exclusion devices are estimated to reduce shrimp trawl related mortality by as much as 94 percent for loggerheads and 97 percent for leatherbacks. Total sea turtle bycatch estimates for 2010 were 6,850 individuals, 6,199 of which were estimated to be mortalities in the two southeast shrimp trawl fisheries (NMFS 2013b). The U.S. Gulf of Mexico fishery had an estimated bycatch mortality of 5,166 individuals (18 leatherback, 778 loggerhead, and 486 green and 3,884 Kemp’s ridley sea turtles). The Southeastern Atlantic fishery had an estimated bycatch mortality of 1,033 individuals (8 leatherback, 673 loggerhead, and 28 green and 324 Kemp’s ridley turtles). The fishery with the next highest estimated sea turtle bycatch, particularly of leatherbacks and loggerheads, is the Atlantic highly migratory species pelagic longline fishery. From 1999 to 2003, the fleet interacted with an average of 772 loggerhead and 1,013 leatherback sea turtles per year, based on observed takes and total reported effort (NMFS 2015). Sea turtle bycatch in the Atlantic pelagic longline fishery has decreased significantly in the last decade. In 2005, the fleet was estimated to have interacted with 275 loggerhead and 351 leatherback sea turtles outside of experimental fishing operations. These numbers have been further reduced to 259 loggerhead and 268 leatherback sea turtles interactions in 2014 (NMFS 2015).

Five other fisheries had a combined estimated sea turtle bycatch of 133.4 individuals in 2010 (live and dead): the U.S. Gulf of Mexico reef fish bottom longline fishery (26.5 loggerhead turtles), U.S. Gulf of Mexico reef fish vertical line fishery (32.9 loggerhead turtles), large coastal and small coastal shark aggregates (drift, strike, and bottom gillnet; 2.9 Kemp’s ridley turtles and 8.9 loggerhead turtles), Southeastern Atlantic snapper-grouper vertical line fishery (56.3 green turtles), and the Southeastern Atlantic and Gulf of Mexico shark bottom longline fishery (5.8

loggerhead turtles) (NMFS 2013b). The Atlantic sea scallop fishery also results in sea turtle bycatch, primarily loggerhead (estimated 49 captured in 2013).

7.1.4 Aquatic Invasive Species

Aquatic invasive species are aquatic organisms introduced into new habitats and subsequently produce harmful impacts on the natural resources in and human uses of these ecosystems (http://www.anstaskforce.gov). Not all non-native (also called alien or nonindigenous) species are considered invasive. Overall, there have been 374 documented invasive species in U.S. waters, 150 of which have arrived since 1970 (Pew 2003). The Nonindigenous Aquatic Species database[6] lists 53 non-native species reported in Florida's brackish and marine waters. Among these are 16 species with established populations in one or more of the estuaries and coastal areas of Florida. The presence of established populations for 32 of the species is unknown and five species failed to establish populations. Many of the fish species are aquarium releases and some of the established populations were actually stocked as forage fish (e.g., shad and blueback herring). The lionfish, originally from the indo-pacific is a particularly harmful invasive fish species in Florida's waters. Lionfish are a major predator on commercial and sport fish species and the herbivorous fish species that are important to controlling algal growth on coral reefs (Lesser and Slattery 2011, Albins and Hixon 2013, Cote et al. 2013). Their presence in reef systems has been associated with severe declines in fish abundance (Albins and Hixon 2008). Initial observations in the mid-1980s are attributed to aquarium releases. They are established in coastal waters from North Carolina to South America. Lionfish have invaded the Loxahatchee estuary (i.e., Jupiter Inlet on the Atlantic coast of Florida). Over 200 young-of-year individuals ranging from 23 to 185 mm were collected over a one-year survey period. They were primarily associated with man-made structures and associated debris along the shoreline as far as 5.5 km inland (Jud et al. 2011).

Introduced aquatic invasive species are one of the main sources of risk to ESA-listed species, second only to habitat loss (Wilcove and Chen 1998). They have been implicated in the endangerment of 48 percent of the species listed under ESA (Czech and Krausman 1997). The USFWS considers invasive species to be a significant contributing factor in determining the “threatened” or “endangered” status of many native species (OTA 1993, Ruiz et al. 1997). Invasive species affect aquatic environments in many different ways. They can reduce native species abundance and distribution, and reduce local biodiversity by out-competing native species for food and habitat. They may displace food items preferred by native predators, disrupting the natural food web. They may alter ecosystem functions. Exotic plants can clog channels and interfere with recreational fishing and swimming. Introduced non-native algal species combined with nutrient overloading may increase the intensity and frequency of algal

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[6] These data are preliminary or provisional and are subject to revision. They are being provided to meet the need for timely best science. The data have not received final approval by the U.S. Geological Survey (USGS) and are provided on the condition that neither the USGS nor the U.S. Government shall be held liable for any damages resulting from the authorized or unauthorized use of the data.

blooms. An overabundance of algae can lead to depleted DO. Oxygen depletion can result in "dead zones," murky water, seagrass and coral habitat degradation, and large-scale fish kills (Deegan and Buchsbaum 2005).

Harmful Algal Blooms

Florida monitors for HABs in fresh, estuarine, and marine waters. Blooms can occur any time of year in Florida, due to its subtropical climate. The HABs are caused by a suite of unique taxa that can bloom under particular physical, chemical, and biological conditions. The drivers of some HABs are well understood, while the drivers of other HABs, such as the red tide organism *Karenia brevis*, are still unclear. While HABs can occur naturally, they are frequently associated with elevated nutrient concentrations. HABs may produce toxins that contaminate shellfish or finfish, making them unsuitable for human consumption. They can also affect plant and animal communities. The Gulf of Mexico Alliance, a partnership between Alabama, Florida, Louisiana, Mississippi, and Texas, is working to increase regional collaboration to enhance the Gulf's ecological and economic health. Reducing the effects of HABs is one of its water quality priorities.

Freshwater cyanobacteria (or blue-green algae) blooms have received increased attention in recent years because of their potential to produce toxins that can harm humans, livestock, domestic animals, fish, and wildlife. While blooms of cyanobacteria can occur naturally, they are frequently associated with elevated nutrient concentrations, slow-moving water, and warm temperatures. Cyanotoxins are bioactive compounds naturally produced by some species of cyanobacteria that can damage the liver (hepatotoxins), nervous system (neurotoxins), and skin (dermatotoxins) of humans and other animals. Potentially toxigenic cyanobacteria have been found statewide in Florida's rivers, streams, lakes, and estuaries. There are also concerns that freshwater cyanotoxins can be transported into coastal systems. The results of the Cyanobacteria Survey Project (1999–2001), managed by the Harmful Algal Bloom Task Force at the FWCC Fish and Wildlife Research Institute, indicated that the taxa *Microcystis aeruginosa, Anabaena* spp., and *Cylindrospermopsis raciborskii* were dominant, while species with the genera *Aphanizomenon, Planktothrix, Oscillatoria*, and *Lyngbya* were also observed statewide but not as frequently. Cyanotoxins (microcystins, saxitoxin [STX], cylindrospermopsins, and anatoxin) were also found statewide (Williams et al. 2007). Other cyanobacteria of concern in Florida are reported in Abbott et al. (2009).

More than 50 marine and estuarine HAB species occur in Florida and have the potential to affect public health, water quality, living resources, ecosystems, and the economy. Any bloom can degrade water quality because decomposing and respiring cells reduce or deplete oxygen, produce nitrogenous byproducts, and form toxic sulfides. Declining water quality can lead to animal mortality or chronic diseases, species avoidance of an area, and reduced feeding. Such sublethal, chronic effects on habitats can have far-reaching impacts on animal and plant communities. *Karenia brevis*, sometimes mixed with related *Karenia* species, causes red tides that are an ongoing threat to human and environmental health in the U.S. Gulf of Mexico.

Blooms occur annually on the west coast of Florida and less frequently in the Panhandle and east coast. *Karenia brevis* produces brevetoxins that can kill fish and other marine vertebrates, including manatees, sea turtles, and seabirds. Blooms of the STX-producing dinoflagellate *Pyrodinium bahamense* have been linked to the bioaccumulation of the neurotoxin STX in puffer fish and more than 20 cases of saxitoxin puffer fish poisoning in Florida (Landsberg et al. 2006). While these blooms raise serious concerns about the ecology of affected ecosystems, there have not been any wide-scale animal mortality events attributed to STXs in Florida. As a tropical species, *P. bahamense* has seldom bloomed north of Tampa Bay on the west coast or north of the Indian River Lagoon on the east coast. Blooms are generally limited to May through October (Phlips et al. 2006). In Florida, *Pyrodinium* is most prevalent in flow-restricted lagoons and bays with long water residence times and salinities between 10 and 30 practical salinity units. The latter conditions competitively favor *Pyrodinium* because of its slow growth rates and euryhaline character (Phlips et al. 2006). Blooms also appear to be accentuated during periods of elevated rainfall and nutrient loads to lagoons (Phlips et al. 2010), suggesting a link between coastal eutrophication and the intensity and frequency of blooms. However, discharges of naturally tannic waters from wetlands during high-rainfall events can also produce favorable conditions for this organism. These observations also point to the potential role of future climate trends in defining the dynamics of HAB species in Florida (Phlips et al. 2010).

Other bloom-forming marine species can be divided into two categories: toxin-producing species and taxa that form blooms associated with other problems, such as low oxygen concentrations, physical damage to organisms, and general loss of habitat. Potential toxin-producing planktonic marine HAB species include the diatom group *Pseudo-nitzschia* spp.; the dinoflagellates *Alexandrium monilatum, Takayama pulchella, K. mikimotoi, K. selliformis, Karlodinium veneficum, Prorocentrum minimum, P. rhathymum*, and *Cochlodinium polykrikoides*; and the prymnesiophytes *Prymnesium* spp. and *Chrysochromulina* spp., and the raphidophyte *Chattonella* sp. (Abbott et al. 2009). Many of these species are associated with fish or shellfish kills in various ecosystems around the world (Landsberg 2002). Additionally, benthic cyanobacteria and macroalgae blooms have been observed on Florida’s coral reefs and have been associated with mortality and disease events involving various organisms (Lapointe et al. 2004, Paul et al. 2005, Richardson et al. 2007).

Although many HAB species have been observed at bloom levels in Florida (Phlips et al. 2011), uncertainty remains over the relative toxicity of the specific strains. In addition to ichthyotoxic HAB species that directly cause fish kills, the list of HAB species linked to hypoxia or other density-related issues (e.g., allelopathy, physical damage to gills of fish) is extensive and includes almost any species that reaches exceptionally high biomass. Examples include the widespread bloom-forming planktonic dinoflagellate *Akashiwo sanguinea*, in the Indian River Lagoon and the St. Lucie Estuary, and the cyanobacterium *Synechococcus* in Florida Bay (Phlips et al. 2010, Phlips et al. 2011). Many fish kills, particularly those occurring in the early morning hours, are due to low DO levels in the water associated with the algal blooms and are not necessarily the result of toxins.

Another important issue associated with HABs is the loss or alteration of overall habitat quality. Prolonged and intense coastal eutrophication can result in domination by a select few species, resulting in a loss of diversity and alteration of food web structure and function. For example, during major *Pyrodinium* blooms, 80 percent to 90 percent of total phytoplankton biomass is attributable solely to this species (Phlips et al. 2006). Similar domination by a single species occurs in benthic ecosystems, where massive blooms of green and red macroalgae have periodically over-run some shallow habitats of the Florida coast (Lapointe and Bedford 2007).

7.1.5 Aquatic Impairments

The year 2010 is the most recent EPA-approved 303(d) list of impaired waters for Florida found in the Assessment and TMDL Tracking and Implementation System database (Table 3[7]), The 2010 data indicate 613 waterbody segments on the CWA 303(d) impaired list due to excess nutrients, algae, or the nutrient indicator Chl-a. There are 1049 waterbody segments listed as impaired due to DO, 35 of which are listed due to excess oxygen demand. An additional 608 waterbody segments are listed as impaired with coliform bacteria and 19 waters are listed due to un-ionized ammonia, suggesting excess nutrient loading in these areas.

Table 3. Waters Listed as Impaired for Nutrients or Nutrient-Related Measures.

Cause of Impairment	EPA 2010	Added by Florida 2012 and 2013	Removed by Florida 2012 and 2013	Net count of impaired waters
Chl-a	321	46	134	233
Excess Algal Growth	29	21	9	41
Phosphorus, Total	6	-	-	6
Trophic State Index	257	26	24	259
DO (Biochemical Oxygen Demand)	35	25		60
Pathogens (Coliform)	608	120	96	632
Unionized Ammonia	19	2	7	14

A more recent list of impaired and delisted waters is found on the Florida website. This list includes data up to 2013 that is not yet integrated into EPA's 303(d) list of impairments. The "Added by Florida 2012-2013" column in Table 3 indicates how many additional waters were identified as impaired and the "Removed by Florida 2012-2013" column identifies how many waters were delisted. Among assessed waters, about 8 percent were delisted because they were found to be unimpaired or incorrectly assessed, 3.7 percent were delisted because a TMDL was adopted to address the impairment, and 4.4 percent were delisted due to the re-assignment, retirement, or realignment of the water body identification number, thereby changing the specific standard by which it was assessed.

The April 2014 Integrated Water Quality Assessment for Florida acknowledged that new WQS NNC had been adopted and indicated that final adoption occurred after the period covered by the

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[7] accessed 12/14/2015 at http://iaspub.epa.gov/apex/waters/f?p=ASKWATERS:V_WO_CURRENT_IMPAIRMENTS_LIST:::::P4_OWNER:ATTAINS

assessment period, and therefore were not to assess attainment in this report. Trends analysis for 38 river stations showed increases in one or more indicators of nutrients (total Kjeldahl nitrogen, nitrate-nitrite, TP, total organic carbon, and/or Chl-a) at 27 stations. Decreasing DO trends were found at three of these stations and increasing fecal coliform occurred at six of these stations. However, increasing trends for fecal coliform also occurred in four stations that did not show a trajectory towards adverse nutrient or DO conditions. Among the 38 stations, only five had trends indicating stable or improving water quality for the parameters evaluated.

8 EFFECTS ANALYSIS: EXPOSURE AND RESPONSE

The exposure analysis characterizes the spatial extent and intensity of stressors associated with an action and the overlap of that exposure with ESA-listed species and habitat. Exposure assessment for the NNC is not straightforward because nutrients are not direct stressors. When in excess, nutrients lead to eutrophication and the stressors and adverse effects associated with eutrophication.

The response analysis follows the exposure analysis and is organized along the risk hypotheses described in Section 8.1.2. To review, risk hypotheses are statements that describe the relationships among stressor, exposure, and the environmental values to be protected. The objective of our assessment is to determine whether the NNC approved by EPA would directly or indirectly adversely affect individual survival or fitness such that the extinction risk of ESA-listed populations or species would be increased or that designated critical habitat necessary for the persistence of ESA-listed species would be adversely affected. The values to be protected are therefore the survival and fitness of individuals and the value of designated critical habitat for conservation of an ESA-listed species. Risk hypotheses constructed in Section 0 placed information on the water quality parameters for which EPA has approved standards in context of species and designated critical habitat attributes that may be affected by those parameters.

8.1 Problem Formulation

As described in the *Approach to the Assessment*, the problem formulation in this opinion first characterizes potential stressor pathways for the water quality parameters for which EPA has approved standards. Problem formulation then uses this information to focus the assessment on those aspects of the status of the species and their designated critical habitat and baseline conditions (e.g., natural and human influenced biogeochemistry of Florida ecosystems) that may be affected by the implementation of these standards. Taken together, this information is used to identify appropriate endpoints for the response analysis and construct risk hypotheses that will integrate exposure and response during risk characterization.

8.1.1 Stressors of the Action

The water quality criteria developed by Florida, and approved by EPA, are for natural constituents of surface waters that, when higher or lower than natural levels, can introduce stress and alter aquatic communities. The NNC criteria are intended to prevent adverse water quality conditions in order to support natural populations of aquatic flora and fauna. TP and TN encompass all forms of nitrogen and phosphorus that may be present in a water body. Plants and microbes require nitrogen and phosphorus for growth and reproduction. The nutrient criteria approved by EPA are expressed as TP and TN, which may include dissolved organic, dissolved inorganic, or particulate forms of nitrogen and phosphorus. Transformations among these forms depend on environmental conditions such as pH, temperature, and DO.

The following model and narratives describe fundamental ecological principles of how adverse nutrient conditions can affect aquatic life. The information is adapted from two sources, Dodds (2002) and the stressor profiles found on EPA's Causal Analysis Decision/Diagnostic Information System (http://www3.epa.gov/caddis/ssr_definitions_str.html). The models follow a schema that relates the pathway to adverse biological responses through modes of action, intermediate states, and proximate stressors. Modes of action for these models are processes, such as photosynthesis or sedimentation that result in a change in environmental state. Intermediate states are the conditions, such as increased plant biomass or bedded sediment, resulting from such processes. Proximate stressors are the actual toxicants, physiological stressors, or resource limitations most directly linked to a biological response. The models share a section reflecting the interdependence of species (generalized in Figure 11). Conditions that directly affect organism survival and lifecycle processes (i.e., effects to growth, fecundity, and recruitment) can also affect the survival and lifecycle processes of prey species and the quality of the non-biological habitat properties (e.g., suitable temperature, substrate, water clarity).

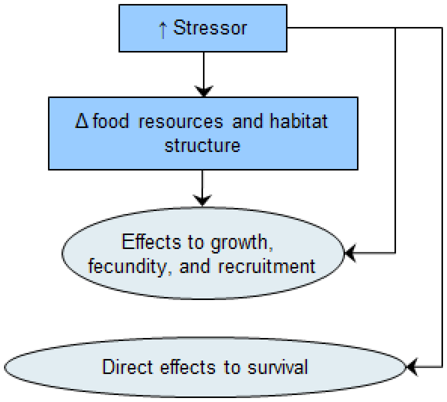


Figure 11. Direct and Indirect Effects of Stressors Due to the Interdependence of Species.

NUTRIENTS

Under natural conditions, essential nutrients contribute to the proper structure and function of healthy ecosystems. However, in excessive quantities, nutrients can have adverse effects on ecosystems, and nutrient enrichment, which leads to eutrophication, often ranked as one of the top causes of water resource impairment (Bricker et al. 2008, USEPA 2014).

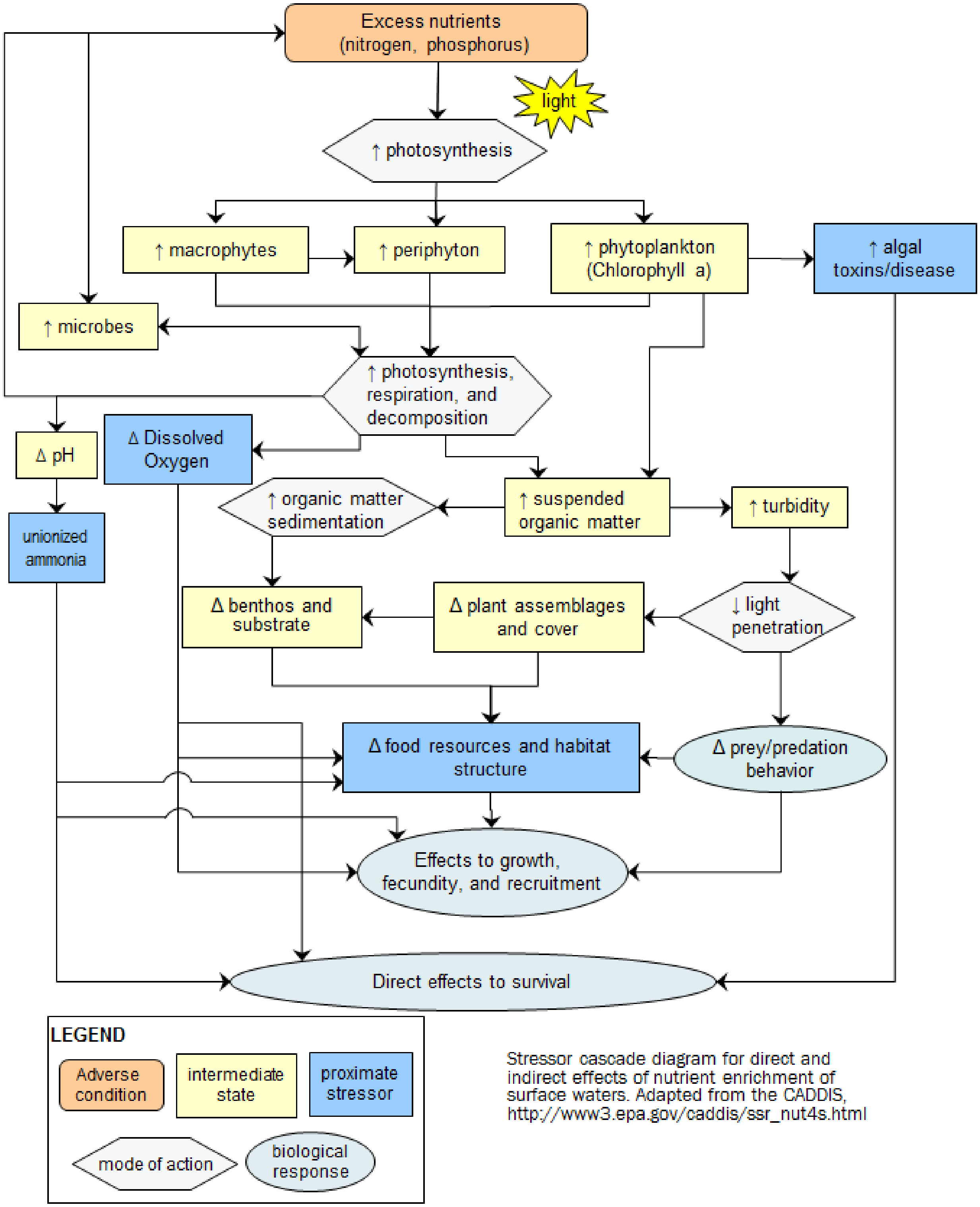
Eutrophication alters the composition and species diversity of aquatic communities through intensifying competition by those species, native or invasive, that are better adapted to eutrophic environments (Nordin 1985, Welch et al. 1988, Carpenter et al. 1998, Smith 1998, Smith et al. 1999). In some cases, it may result in ESA-listed species experiencing increased mortality from competitors. Thus, eutrophication can have cascading effects that change ecosystem structure at numerous trophic levels. Nuisance levels of algae, as indicated by Chl-a levels, and other aquatic vegetation (macrophytes) can develop rapidly in freshwater and marine habitats in response to nutrient enrichment when other factors (e.g., light, temperature) are not limiting. The relationship between nuisance algal growth and nutrient enrichment has been well-documented (Welch et al. 1992, VanNieuwenhuyse and Jones 1996, Dodds et al. 1997, Chetelat et al. 1999). In addition to outcompeting native aquatic plants for space and light, the proliferation of nuisance algae can lead to the occurrence of harmful algal blooms (e.g., brown tides, toxic *Pfiesteria piscida* outbreaks, some types of red tides) which contain microalgae that produce potent toxins. Symptoms from toxin exposure range from neurological impairment to gastrointestinal upset to respiratory irritation, and sometimes result in severe illness and death (Lopez et al. 2008). In marine systems, algal toxins have caused massive fish kills, along with deaths of whales, sea lions, dolphins, manatees, sea turtles, birds, and wild and cultured fish and invertebrates (Landsberg 2002, Shumway et al. 2003). Eutrophication is believed to be a likely contributor to the increased occurrence of harmful algal blooms (Heisler et al. 2008). In addition to its association with harmful algal blooms and algal toxins, eutrophication has also been linked to increases in bacteria biomass (Carr et al. 2005). Bacteria have been associated with mortality among fish, turtles, and alligators (Shotts et al. 1972). There has been an increase in the number of unusual marine mammal mortality events reported in the U.S. and this is believed to be associated with the increasing occurrence of harmful algal blooms. The timing of the blooms and strandings of marine mammals suggests that species that forage both inshore and offshore can be affected. NOAA’s Marine Mammal Health and Stranding Response Program is finding more mammal stranding events to be linked to biotoxins (Gulland and Hall 2007, de la Riva et al. 2009).

The accumulation of algal biomass through excessive productivity can reduce available habitat, and the decay of this organic matter may lead to reductions in DO in the water, which in turn can cause problems such as fish kills and release of toxic substances or phosphates that were previously bound to oxidized sediments (Chorus and Bartram 1999). High biomass blooms of toxic and nontoxic algae resulting from excess nutrients or eutrophication is a common type of event that can cause hypoxia or anoxia (low or no DO), which suffocates fish and bottom-dwelling organisms and can sometimes lead to hydrogen sulfide poisoning (Lopez et al. 2008). Hypoxia can cause habitat loss, long-term weakening of species, change in species dynamics and even fish kills. Because hypoxia often occurs in estuaries or near shore areas where the water is poorly mixed, nursery habitat for fish and shellfish is often affected. Without nursery grounds the young animals cannot find the food or habitat they need to reach adulthood. This causes years of weak recruitment to adult populations and can result in an overall reduction or destabilization of important stocks. High biomass blooms can also directly inhibit growth of beneficial vegetation by blocking sunlight penetration into the water column (Onuf 1996). For example, an excessive accumulation of filamentous benthic algae or other macrophytes during the peak summer growing season can alter stream flow as well as the availability of benthic habitat for stream invertebrates and vertebrates (Welch et al. 1989, Chessman et al. 1992). Macroalgal blooms reduce sunlight penetration and can overgrow or displace seagrasses and corals as well as foul beaches (Valiela et al. 1997). Bloom-inflicted mortalities can degrade habitat quality indirectly through altered food webs or hypoxic events caused by the decay of dead animals (Lopez et al. 2008).

The stressor-to-response pathways for the direct and indirect effects of excess nutrients are described in Figure 12. Excess nutrients accelerate the production and turnover of plant and algal biomass and alter plant species composition. Aquatic plants and microbes require N and P for growth and reproduction. Given adequate light, photosynthesis converts carbon dioxide into biomass growth of macrophytes, periphyton, and phytoplankton. The consumption of carbon dioxide also generates oxygen and increases water pH. The breakdown of plant and algal biomass is mediated by microbes, which consume oxygen during respiration and release carbon dioxide, lowering pH. Plants also respire and consume oxygen, but photosynthesis during the day generates more oxygen than consumed. When photosynthesis pauses at nighttime, plant respiration and microbial decay continues, resulting in a diurnal cycle of peak DO and pH during daylight hours and lowest DO concentrations and pH observed pre-dawn. In addition, direct toxic effects can occur in waters with elevated nitrogen in the form of ammonia, with the more toxic form ammonium more prevalent at high pH. In addition to the potential for ammonia toxicity under high nutrient loadings, toxins may also be produced by some algae that thrive in eutrophic conditions. Accumulation and increased turnover of algal and plant biomass (i.e., death, decay, nutrient release) generates suspended solids in the form of organic particulates and phytoplankton, contributing to turbidity from natural and human-caused erosion and sediment resuspension. Increased turbidity affects light penetration into the water column and the ability of aquatic plants to photosynthesize and survive and the effectiveness of sight-dependent behaviors such as foraging by sight feeders, reproductive displays, and predator evasion. This, in turn, affects the degree of coverage of the substrate by plants and benthic organisms that are reliant on plants. Suspended organic matter can eventually accumulate upon and smother plants, animals, and benthic habitat surfaces. Increases in plant and microbial biomass or productivity may result in negative ecological effects by:

- altering food resources: the amount and type of food resources or their palatability (e.g., changes in algal cell size affects filter-feeding animals);
- increased microbial infection of invertebrates or fish;
- altering habitat: light penetration, diurnal DO cycle, changes in benthic interstitial space, availability of macrophytes as habitat;

- stimulating generation of toxins: some algae that thrive in eutrophic conditions can be toxic to fish and invertebrates;
- increasing mortality through favoring nitrogen pathways increasing the formation of toxic unionized ammonia; and
- changes in community structure, even without overall increases in primary producers, due to alterations of nutrient availability ratios.



Stressor cascade diagram for direct and indirect effects of nutrient enrichment of surface waters. Adapted from the CADDIS, http://www3.epa.gov/caddis/ssr_nut4s.html

Figure 12. Direct and Indirect Effects of Excess Nutrients on Biological Responses.

While Figure 12 provides the conceptual pathway for TN and TP to promote eutrophic conditions, nutrient and biological community dynamics in shallow, subtropical estuarine systems such as those evaluated in this opinion are complex, with phytoplankton production also influenced by the interactions of physical factors affecting the proportion of nutrients biologically available for growth, the ability of phytoplankton to assimilate those nutrients, the species of phytoplankton present, and abundance of phytoplankton grazers. These factors include salinity, temperature, light that are, in turn, influenced by diurnal cycles, annual cycles, and seasonal changes in organic inputs (Montgomery et al. 1991, Havens et al. 2001, Paerl et al. 2010, Phlips et al. 2015, Filippino et al. 2017). Additionally, events such as storms and the normal operation, or failures, of flood control structures increase freshwater inputs, decrease salinity, altering the dynamics of the phytoplankton/grazer communities, in particular favoring proliferation of blue-green algae (Montgomery et al. 1991, Lammers et al. 2013, Filippino et al. 2017, Lemley et al. 2017). In cases where algal blooms are occurring or have occurred, association of TN and TP concentrations with phytoplankton distribution (i.e., Chl-a measures) is disrupted through drift of algae and temporal lag in responses to nutrient loading (Malchow 1993, Malchow and Shigesada 1994, Schernewski et al. 2005, Serizawa et al. 2009, Amemiya et al. 2012, Phlips et al. 2015) and the dissipation/depletion of nutrients as they are consumed under bloom conditions (Paerl 1988).

8.1.2 Risk Hypotheses for Evaluating Approved Criteria and Limits

Risk hypotheses are statements that organize an analysis by describing the relationships among stressor, exposure, and the environmental values to be protected (also referred to as the assessment endpoints). The objective of this opinion’s assessment, per the ESA, is to determine whether the WQS approved by EPA would directly, or indirectly, adversely affect individual survival or fitness such that the continued existence of ESA-listed populations or species would be jeopardized, or that designated critical habitat necessary for the conservation of ESA-listed species would be adversely modified or destroyed. Generally speaking, the values to be protected are therefore the survival and fitness of individuals and the value of designated critical habitat for conservation of an ESA- listed species. Risk hypotheses are constructed by placing information on the water quality parameters for which EPA has approved standards in context of species and designated critical habitat attributes potentially affected by those parameters.

Nutrients directly affect photosynthesizing organisms through stimulating photosynthesis. Both photosynthetic and non-photosynthetic organisms are indirectly affected when excess nutrients alter the resources and physical properties of habitats through accelerated accumulation and turnover of plants and algae and the consequent cascading changes in aquatic chemistry and organic matter (i.e., eutrophication). The Florida NNC are intended to remediate or prevent eutrophication, so these criteria are not presumed to result in eutrophication. The analysis needs to determine whether the criteria are adequately protective and prevent eutrophication. If the criteria are not protective and support eutrophication, the adverse effects of stressors resulting from eutrophic conditions must be evaluated for ESA-listed species and designated critical habitat. For this reason, a tiered approach first determines whether the NNC will promote eutrophic conditions before applying risk hypothesis to evaluate the potential for adverse effects associated with eutrophic conditions (Figure 12). The exposure analysis will evaluate the potential for eutrophication prior to establishing the distribution and overlap of those areas where the NNC support eutrophication with ESA-listed resources under NMFS' jurisdiction. In cases where the NNC support eutrophication, the opinion will address the following risk hypotheses, as appropriate:

- NNC will support eutrophic conditions that affect the survival and fitness of individuals through:
    - lethal and sublethal exposures to ammonia
    - lethal and sublethal exposures to algal toxins
    - lethal and sublethal effects of DO extremes
    - lethal and sublethal infections
    - lethal and sublethal smothering
    - altered light penetration/turbidity
    - altered colonization substrate

- NNC will support eutrophic conditions that have indirect effects to survival and fitness through:
    - reduction in extent of habitat
    - reduction in prey species

The individual risk hypotheses scenarios identified above do not necessarily apply to each species addressed in this opinion. For example:

- North Atlantic right whale and sea turtles breathe air, therefore these species will not be directly affected by DO concentrations in water; and
- coral species that are supported largely through photosynthesis would be affected by changes in light penetration, but not necessarily by reductions in planktonic prey species.

In short, a risk hypothesis is assessed for a species only if it is applicable to that species. These considerations are summarized for the species groups in Table 4. This table provides the organizing framework for the effects analysis of this opinion. Where the table contains a check mark (✓), the risk hypothesis applies to the species in question. Where there is not a check mark, text in the table explains why the hypothesis is not applicable to that species and the stressor scenarios are therefore determined to be “Not Applicable” for these ESA-listed species. Applicability of a risk hypothesis does not in itself indicate a conclusion that such adverse effects are expected to occur, but instead merely indicates that it is a hypothesis that should be evaluated.

In addition to assessing effects on species, this opinion also evaluates effects to designated critical habitat. Designated critical habitat in Florida’s waters is designated for the North Atlantic right whale, the loggerhead sea turtle, elkhorn and staghorn corals, smalltooth sawfish, gulf sturgeon, and Johnson’s seagrass.

The designated critical habitat assessment evaluates the effects of the NNC levels that were approved as WQS on the value of the designated critical habitat to the conservation of the species, with a focus on the physical and biological features of designated critical habitat essential to the conservation of the species. The overarching risk hypothesis is:

> *"Florida's NNC, approved by EPA, support conditions that adversely affect the critical habitat, including the features that are essential to the conservation of the species."*

Table 5 lists the physical and biological features specified in critical habitat designations for each species and relates these to the designated critical habitat risk hypotheses. The designated critical habitat analysis also revisits the indirect effects described in Table 4 on values that are not specified in the critical habitat designation, but may occur within the spatial extent of designated critical habitat, to the extent effects on such values would affect the value of the habitat to the conservation of the species.

Table 4. Applicability Determinations for Environmental Protection Agency approved Numeric Nutrient Criteria Risk Assessment Hypotheses for Endangered Species Act-listed Species under Jurisdiction of the National Marine Fisheries Service.

	North Atlantic right whale	Green, hawksbill, Kemp's ridley, leatherback, and loggerhead sea turtles	Atlantic and shortnose sturgeon[a], smalltooth sawfish, and Nassau grouper	The Acropora, the Orbicella pillar coral, and rough cactus coral	Johnson's seagrass
Hypothesis: NNC will support eutrophic conditions that affect the survival and fitness[b] of individuals through:					
lethal and sublethal exposures to ammonia	**No:** breathes air, drinks little to no seawater	**No:** Drinks seawater but not affected even at high concentrations	✓	✓	✓
lethal and sublethal exposures to algal toxins	**No:** does not forage in Florida waters, drinks little to no seawater	✓	✓	✓	**No:** Plants are not known to be susceptible to algal toxins
lethal and sublethal DO extremes	**No:** breathes air	**No:** breathe air	✓	✓	✓[c]
lethal and sublethal infections	✓	✓	✓	✓	**No:** Eutrophication is not known to contribute to seagrass disease
lethal and sublethal smothering by algae	**No:** mobile	**No:** mobile	**No:** mobile	✓	✓
altered turbidity/light penetration[d]	**No:** not light dependent	**No:** not light dependent	**No:** not light dependent	✓	✓
Hypothesis: NNC will support eutrophic conditions that have indirect effects to survival and fitness through:					
reduction in extent of habitat/altered substrate	✓	✓	✓	✓	✓
reduction in prey species	**No:** does not forage in FL waters	✓	✓	✓	**No:** autotrophic

[a] St. Marys River identified as a spawning river in proposed critical habitat for Atlantic sturgeon because young-of-year fish were captured in this river in 2014.

[b] Reproductive output, colonization, or offspring viability.

[c] While seagrass generates oxygen, the species is vulnerable to anoxic extremes when coupled with reduced light penetration resulting from algal blooms under eutrophic conditions

[d] Interpreting light penetration/turbidity data with respect to nutrient enrichment is complicated by the need to differentiate the contribution of algae from that of suspended sediment.

Table 5. Applicability Determinations for the Risk Hypothesis: "Florida's Numeric Nutrient Criteria, Approved by the Environmental Protection Agency, Support Conditions that Adversely Affect the Critical Habitat, Including the Features that are Essential to the Conservation of the Species."

Species	Essential Physical and Biological Features	Implications of Standards
North Atlantic right whale	Water of particular depth and temperature, Abundant prey resources, oceanographic features that aggregate prey; Waters free of obstruction and disturbance to allow whales to rest, travel, feed, breed, birth, and raise calves safely	**No:** Not relatable to nutrients, right whales do not feed in FL
Loggerhead turtle Northwest Atlantic Ocean	Nearshore Reproductive Habitat; Nearshore waters directly off the highest density nesting beaches and their adjacent beaches as identified in 50 CFR 17.95 (c) to 1.6 km (1 mile) offshore; Waters sufficiently free of obstructions or artificial lighting to allow transit through the surf zone and outward toward open water; Waters with minimal manmade structures that could promote predators (i.e., nearshore predator concentration caused by submerged and emergent offshore structures), disrupt wave patterns necessary for orientation, and/or create excessive longshore currents	**No:** Not relatable to nutrients
	Breeding Habitat; High densities of reproductive adults; Proximity to primary Florida migratory corridor; and Proximity to Florida nesting grounds.	**No:** Not relatable to nutrients
	Migratory Habitat; Constricted continental shelf area relative to nearby continental shelf waters that concentrate migratory pathways; and; Passage for migration to and from nesting, breeding, and/or foraging areas.	**No:** Not relatable to nutrients
	Sargassum Habitat; Convergence zones, surface-water downwelling areas, the margins of major boundary currents (Gulf Stream), and other locations where there are concentrated components of the Sargassum community in water temperatures suitable for the optimal growth of Sargassum and inhabitance of loggerheads; Sargassum in concentrations that support adequate prey abundance and cover; Available prey and other material associated with Sargassum habitat including, but not limited to, plants and cyanobacteria and animals native to the Sargassum community such as hydroids and copepods; and; Sufficient water depth and proximity to available currents to ensure offshore transport (out of the surf zone), and foraging and cover requirements by Sargassum for post-hatchling loggerheads, i.e., >10 m depth.	**Applicable:** NNC potential effects to predation and prey species
Smalltooth sawfish	Juvenile Nursery Habitat; Red mangroves and adjacent shallow euryhaline habitats due to their function of providing refugia and diverse and abundant forage that facilitate recruitment of juveniles into the adult population	**Applicable:** Potential effects to refugia and forage
Elkhorn coral Staghorn coral	Substrate of suitable quality and availability to support successful larval settlement and recruitment, and reattachment and recruitment of fragments	**Applicable:** Potential effects to substrate
Johnson's seagrass	Adequate water quality, salinity levels, water transparency and Stable, unconsolidated sediments free from disturbance	**Applicable:** NNC potential effects to light penetration substrate

8.2 Exposure

Florida's NNC are intended to prevent eutrophication and promote healthy conditions. The indicators representing healthy conditions include seagrass metrics, Chl-a concentrations, and DO regime. Seagrass metrics are used as indicators of estuary health worldwide, with declines in spatial extent, density, and biomass integrating the influences of multiple stressors (Roca et al. 2016). Chl-a is a useful indicator of plankton growth which integrates nutrient loading in an aquatic system. Overstimulation of photosynthesis and increased algal growth by excess nutrients elevates Chl-a levels above natural conditions (Harding et al. 2014). The DO regime of a system is also an indicator of eutrophic conditions, with photosynthesizing algal biomass elevating DO to supersaturated levels (i.e., DO saturation>100 percent) in daylight and the oxygen consumption processes of respiration and algal decomposition and decay depleting DO at night time (Wenner et al. 2004, Prasad et al. 2011).

The NNC for TP and TN are nutrient levels at which Florida expects these indicators will meet *thresholds* reflecting the reference conditions expected to protect aquatic life (i.e., sufficient seagrass coverage and water clarity, natural Chl-a levels, and natural DO regime). Reference conditions were based on either reference period or reference site data. Reference period data are selected from a period when the water itself was determined to be biologically unimpaired and supporting its most sensitive designated uses. When reference period data were not available, data from an unimpaired, adjacent, and functionally similar reference site, a reference system, were used to represent reference conditions. Once the reference period or reference system was identified, distributional statistics were used to set the TN and TP NNC at a level that to maintain the distribution of reference condition monitoring data, while accounting for natural temporal variability. In contrast, the Chl-a NNC are based on reference conditions meeting the following metrics for each estuary (FDEP 2015a):

- Site-specific seagrass depth (Zc) and water clarity (Kd) targets to achieve 20 percent of surface light at the mean depth of the deep edge of seagrass beds, relative to mean sea level, based on historical or recent seagrass coverages (where available, as proposed by (USEPA 2012b) or by a water management district), using Secchi depth measurements.
- A Chl-a target to prevent nuisance algal blooms (Chl-a not to exceed 20 μg/L) >10 percent of the time based on annual data).
- DO targets to protect aquatic life (including a minimum allowable daily DO saturation of 42 percent, 90 percent of the time, based on annual data).

NMFS concludes, after review of supporting technical documents (USEPA 2012b, FDEP 2015a) supporting the determination of estuary Chl-a NNC that it is reasonable to expect the Chl-a NNC to represent reference (i.e., non-eutrophic) conditions and may therefore be used as an indicator to evaluate whether the TN and TP NNC support eutrophic conditions.

The exposure assessment must determine whether any of the NNC promote eutrophication before determining whether ESA-listed species or designated critical habitat are exposed to the adverse effects of eutrophication. The response analysis will evaluate the potential responses of ESA-listed species that occur in areas where the NNC are expected to promote eutrophication.

Florida's NNC for estuaries are Hierarchy 1, site-specific standards. The NNC include concentration and load-based criteria that were derived through different strategies determined by the quantity and comprehensiveness of available data, and, in some cases, the complexity of the system to be protected. The concentration-based NNC are expressed in arithmetic means or, in cases where the underlying data were positively skewed, geometric means. Geometric means are used to attenuate the short-term variability reflected by skewed data to provide a more reliable long-term estimate of the nutrient status. The arithmetic and geometric mean calculations produce equivalent results when applied to data that are not skewed.

8.2.1 Load-Based Numeric Nutrient Criteria

The NNC that are expressed as loads, that is in terms of mass per unit time, are from existing TMDLs (i.e., nutrient-impaired systems affected by legacy nutrients and vulnerable to multiple factors influencing Chl-a, see Section 8.1.1). These are considered numerical interpretations of Florida’s previous overarching narrative criterion of "no imbalance in natural populations of aquatic flora or fauna." That is to say, if that surface water meets the concentrations of TN, TP, and Chl-a specified by numeric criteria, there will be no imbalance in natural populations of aquatic flora or fauna (i.e., the presence and abundance of species indicative of a healthy ecosystem). If these TMDL analyses have already been developed or as they become developed in the future, they become the applicable criteria for the specific waterbody.

Fifteen of the segment specific proposed NNC are load-based standards for TN and TP based on TMDLs among six estuaries (Table 6). The load-based NNC are applied to areas already known to be nutrient impaired, and recovery from nutrient impairments would be indicated by Chl-a levels that are consistent with Chl-a NNC for these segments, in addition to increased water clarity, DO, and improvements in biological condition.

Table 6. Estuary Segments with Load-based Nutrient Criteria.

Estuary	Segment
Big Bend and Apalachee Bay	Fenholloway River Estuary
Indian River Lagoon System	Banana River Lagoon Central Indian River Lagoon Newfound Harbor North Indian River Lagoon Sebastian River Estuary Sykes Creek Estuary
ICWW	Palm Coast
Lower St. Johns River	Lower St. Johns River and Tributaries
Springs Coast (Crystal River to Anclote River)	Kings Bay
St. Lucie Estuary	Upper North Fork St. Lucie River Upper South Fork St. Lucie River Lower North Fork St. Lucie River Lower South Fork St. Lucie River St. Lucie Estuary

DO THE LOAD-BASED ESTUARY NUMERIC NUTRIENT CRITERIA PROMOTE OR SUSTAIN EUTROPHIC CONDITIONS?

Interpreting the Chl-a observations from segments with TMDL-based NNC requires the assumption that the TMDL limits are implemented and effective. TMDLs establish allowable loadings in order to restore impaired waters to meet standards for the protection of human and aquatic live by reducing pollutant loadings. A Basin Management Action Plans (BMAP) outlines the collaborative efforts of stakeholders to identify current and planned management actions to achieve pollutant load reductions required by the TMDL. A BMAP represents a comprehensive set of strategies: permit limits on wastewater facilities, urban and agricultural best management practices, conservation programs, control of legacy pollutants, and other considerations designed to implement the pollutant reductions established by the TMDL. BMAPS integrate local input and local commitment because they are developed with local stakeholders. They are also enforceable, as they are adopted by Secretarial Order within Florida's Department of Environmental Protection. The effectiveness of BMAPs is evaluated by the stakeholders and Florida's Department of Environmental Protection annually and through a formal re-evaluation every five years. This allows changes to be made to the plan and its actions if goals are not being met. These characteristics of Florida's TMDL program meets each of the characteristics identified by Gross and Hagy (2017) as consistently associated with achieving restoration goals for eutrophic systems:

1. *leadership by a dedicated watershed management agency;*
2. *governance through a bottom-up collaborative process;*
3. *a strategy that set numeric targets based on a specific ecological goal; and*
4. *actions to reduce nutrient loads from all sources*

NMFS interprets evidence of successful TMDL implementation as a trend in decreased TN and TP after the establishment of the TMDL. The TMDLs for these estuary segments were established two to eight years ago. Figure 13 illustrates the data available for these estuary segments, with pre-TMDL data in gold and post TMDL data in blue. The years for which data are available are along the horizontal axis and the concentrations of the nutrient indicators are along the vertical axis of each individual plot in Figure 13. It is important to consider that these data represent a relatively small window in time relative to the amount of time reported for nutrient regimes to shift, given internal (e.g., sediment, biota) nitrogen and phosphorus inputs cycling in the system (Bell et al. 2008, Bell et al. 2014, Riemann et al. 2016, Staehr et al. 2017). Riemann et al. (2016) reviewed recovery of Danish coastal waters following substantial reductions in nitrogen and phosphorus loading in the 1990s. Trends between 1990s and 2013 include overall decline in Chl-a of −0.057 μmol/L/year ($p< <0.0001$), change in water depth for eelgrass growth of −0.006 m/year ($p=0.0080$), and increased macroalgae cover of 0.69 percent/year ($p= 0.0007$). With only one pre-TMDL sampling event, the Fenholloway River Estuary segment does not have useful data for evaluating the load-based NNC. Seven estuary segments evaluated in this opinion have five or less years of post-TMDL data because their TMDL was only recently established or because data for recent years are not available: ICWW Palm Coast, Kings Bay, Newfound Harbor, Sebastian River, St. Lucie River Lower North and South Forks, and Sykes Creek. A coarse pre- and post-TMDL comparison is provided for these estuary segments. Seven to eight years of post-TMDL data are available for the remaining seven estuaries, and these data were used to evaluate whether a trend towards improving conditions can be suggested: Banana River, Central Indian River, Lower St. Johns, North Indian River St. Lucie Estuary, and the St. Lucie River Upper North and South Forks.

Figure 13. Distribution of Observations within Estuary Segments between 1996 and 2016.

Coarse Pre-and Post-Total Maximum Daily Load Comparisons

The pre and post TMDL differences for segments with four or fewer years of post-TMDL data are effectively illustrated in combination violin-box plots (Figure 14).

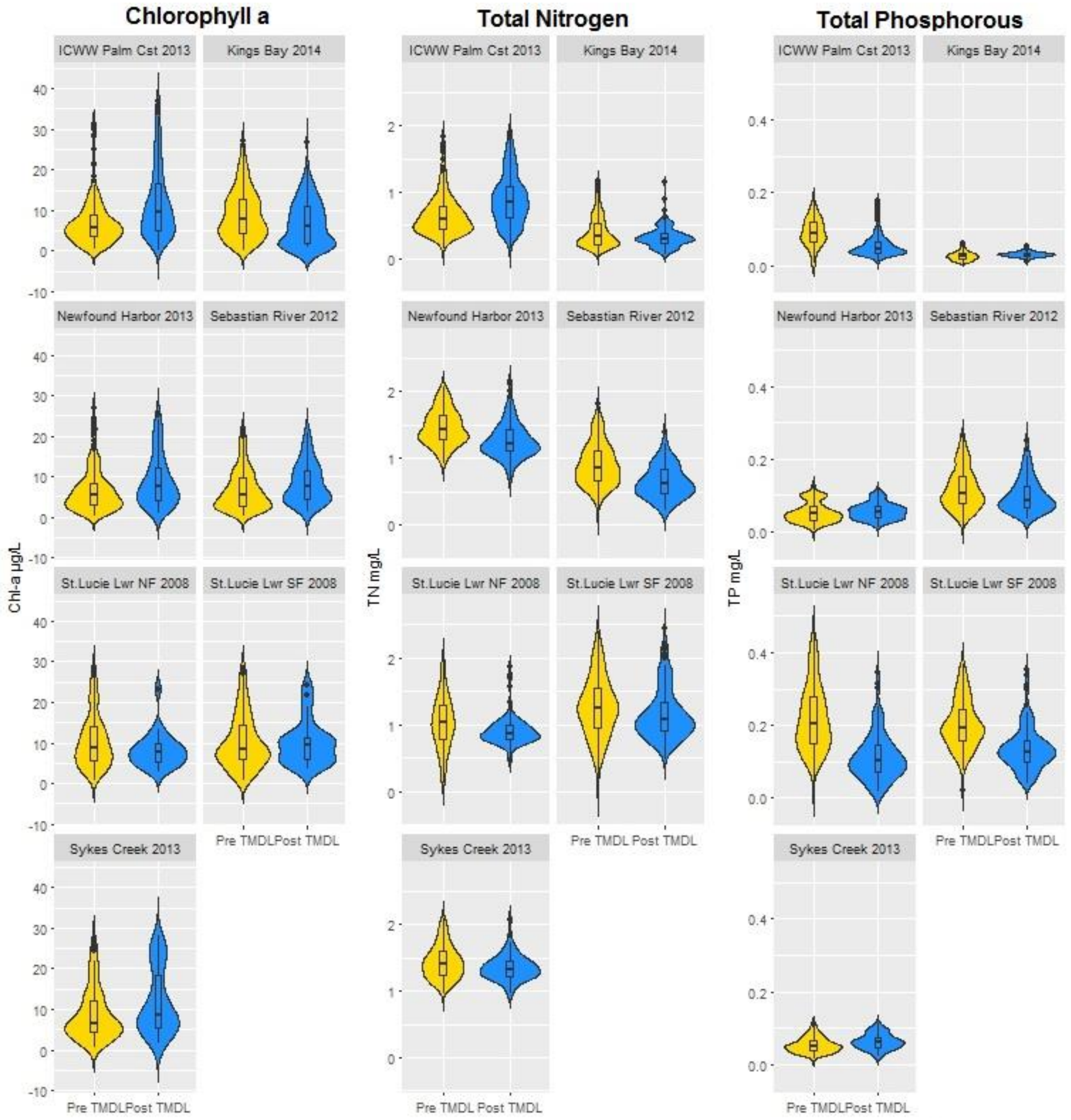


Figure 14. Combination Violin-boxplots Illustrating the Relative Distribution of Pre- and Post-Total Maximum Daily Load Nutrient Response Observations (i.e., Chlorophyll-a) and Nutrient Concentrations (i.e., Total Nitrogen and Total Phosphorous). Violin Width Indicates Observation Frequency.

The violin width indicates the relative number of data points, and the central band of the boxplot within the violin identifies the midpoint of the distribution, with 25 percent of observations occurring above the median band within the box and 25 percent occurring below the median band and within the box. The lines extending from the box account for the remaining lower and upper 25 percent of data and extend to the minimum and maximum observation falling within one standard deviation from the mean. These plots do not include extreme values and outliers because, graphically, they extended the observation range of nutrient indicators, obscuring the distribution of the bulk of the data. Outliers were identified using Tukey's method[8] and removed.

For ICWW Palm Coast, Kings Bay, Newfound Harbor, Sebastian River, and the St. Lucie Lower North and South Forks, post-TMDL improvements in at least one nutrient indicator are suggested, as shown by post TMDL violins with broader bottoms (i.e., greater number of observations at lower nutrient indicator levels) relative to the associated pre-TMDL plots. For example, the violin breadth and median in the plots for the lower North and South Forks of the St. Lucie show more observations at lower concentrations of TN, and TP, post TMDL (blue) relative to pre-TMDL concentrations (yellow).

The pre and post TMDL plots overlap substantially. The absence of a clear difference indicating improved conditions does not necessarily suggest the TMDL loading limits (i.e., the NNC) are too high. Establishment of TMDL limits does not immediately result in compliance in the regulated community and measures taken to reach TMDL goals may be phased to address immediate sources (i.e., point sources) and sources that are more complicated to address, such as legacy nutrient loads and nonpoint sources. Load reductions do not result in immediate changes in the constituent nutrient load biogeochemical cycling (e.g., flux in sediment, biota).

It is important to note that BMAPs have not yet been adopted for ICWW Palm Coast or Kings Bay. The BMAP for Kings Bay is under development. While Florida has not begun to develop a BMAP for ICWW Palm Coast, BMAPs are the primary mechanism through which the state implements TMDLs (see Florida Statutes Subsection 403.067[7]). Meanwhile, Newfound Harbor, Sebastian River, and the St. Lucie Lower North and South Forks are addressed in the Banana River Lagoon (Banana River Lagoon Stakeholders and FDEP 2013), Central Indian River Lagoon (Central Indian River Lagoon Stakeholders and FDEP 2013), and St. Lucie (St. Lucie River and Estuary Basin Technical Stakeholders and FDEP 2013) BMAPs, respectively. Sykes Creek is part of the Indian River Lagoon Basin, specifically the Banana River sub-basin. Sykes Creek was identified as a priority location for muck removal in the 2015 annual progress report for the Banana River BMAP, which includes addressing legacy nutrients in sediments (FDEP 2015b).

Post-Total Maximum Daily Load Trends

While the violin plots described in the previous section provide a coarse overall impression of monitoring data pre- and post-TMDL, plotting of annual means for segments with greater than 5 years of post TMDL data can suggest the presence of a trend (Figure 15). For example, plots of Chl-a annual geometric means do not visually suggest a post-TMDL decline while TN and TP appear to decline in the Lower St. Johns and Upper St. Lucie estuary segments. Meanwhile, plots of annual means for the St. Lucie Estuary proper actually suggest a post TMDL spike in nutrient

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[8] Tukey's method is based on distributional statistics, flagging observations that are above and below 1.5 times the interquartile range (i.e., the central 50% of observations). This approach is not dependent on data normality or directly influenced by the underlying mean and standard deviation of the data.

indicators. Applying regression analyses to test whether a statistically significant trend exists among post-TMDL observations requires first confirming that the underlying distributions of the data meet the statistical requirements for regression: errors distributed normally around the mean and consistent variability among response observations (i.e., Chl-a) along the TN or TP gradient. The data are highly dispersed (Figure 13), resulting in broad standard deviations. Regression analysis of such data, provided assumptions of normally distributed errors around the mean and homogeneity of variance are met, will not produce a predictive linear model. One would expect a low R^2, a statistical metric describing how close observations are to the modeled regression line representing the mean response of Chl-a over the range in TN or TP. However, regression of "noisy" data like this can provide evidence of changes in nutrient conditions over the time elapsed since TMDL establishment.

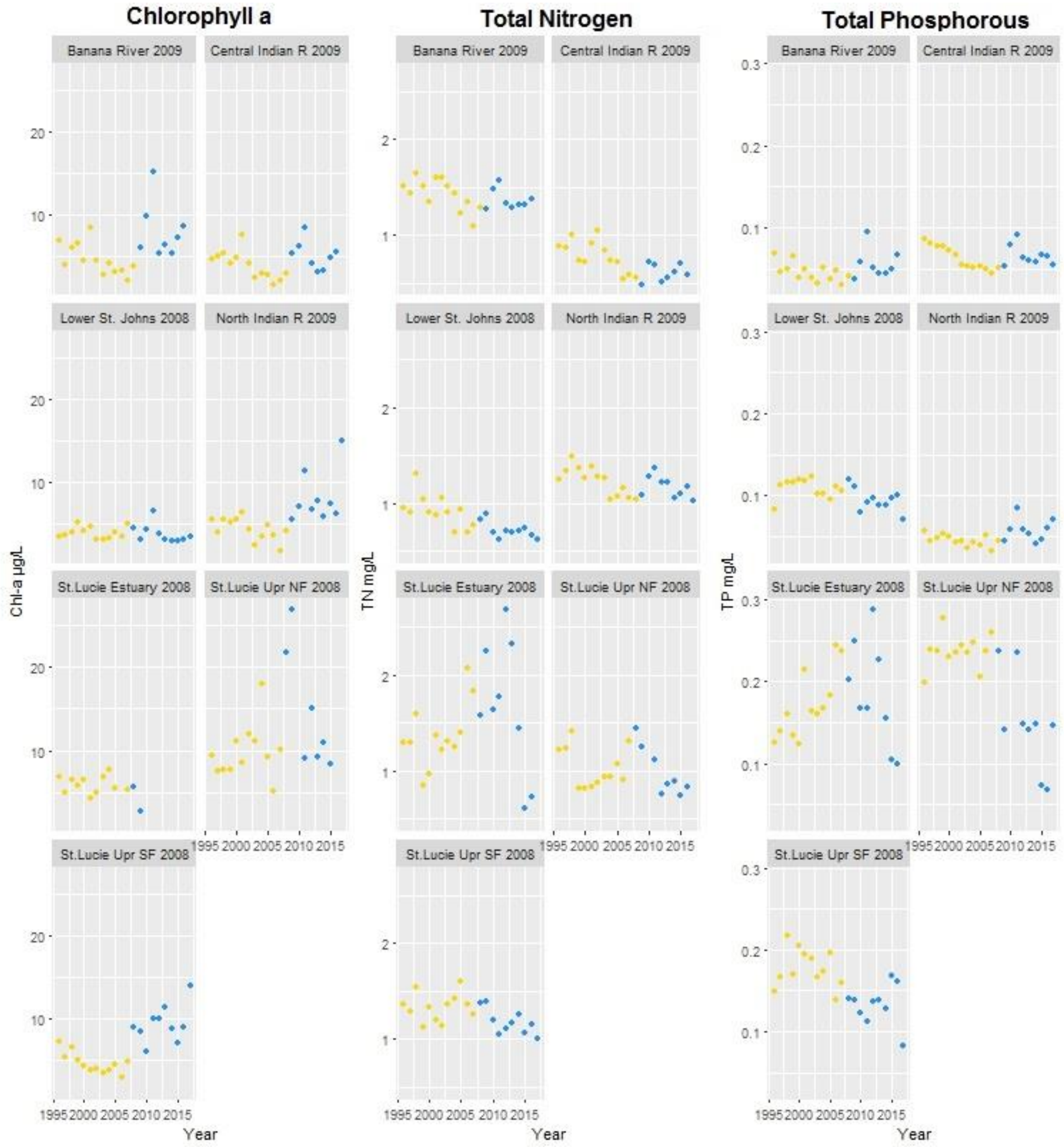

Figure 15. Annual Geometric Means for Monitoring Data between 1996 and 2016.

All seven of the estuary segments with more than 5 years of post-TMDL data had at least one nutrient indicator with data distributions that could be evaluated through simple linear regression. Log transformation of these data was often necessary to meet restrictions for linear regression (Table 7). Regressions of Banana River Lagoon and Central Indian Lagoon, and North Indian River Lagoon data suggested nutrient indicators have increased slightly since

TMDL development. Meanwhile, declines are suggested for TN among the Lower St. Johns, St. Lucie Estuary, and the Upper South Fork of the St. Lucie River.

A BMAP for nutrients in the Lower St Johns estuary segment was adopted in 2008 (Lower St. Johns River TMDL Executive Committee and FDEP 2008). Under the BMAP, nitrogen loading was more than halved within 5 years. The system was within 6 percent of its total TMDL goal in 2013 (FDEP 2013). As of 2016, the BMAP achieved TMDL goals for loadings from wastewater treatment facilities and municipal separate storm sewer systems within the basin. Loadings from non-point stormwater sources and agricultural sources are within 75 percent and 63 percent of their TMDL goals, respectively (FDEP 2016a). Regression analyses confirm reductions in TN. The Chl-a and TP data from this segment were not suitable for regression analysis.

The BMAP for the St. Lucie Basin, which includes St. Lucie Estuary and the Upper South Fork of the St. Lucie River, was adopted in 2013 (St. Lucie River and Estuary Basin Technical Stakeholders and FDEP 2013), so the apparent post-TMDL declines in TN cannot be attributed to implementation of the TMDL through the BMAP. However, the plan noted this decline, describing how annual discharge trends after 2006 resulted in reduced loading from external sources, particularly nutrient loading from Lake Okeechobee.

The Banana River Lagoon, Central Indian River Lagoon, and North Indian River Lagoon are part of the Indian River Lagoon Basin. Nutrient loadings for all three segments are addressed in the same TMDL (FDEP 2009). Algal blooms occurring in these waters are attributed, in part, to legacy nutrient loads in the sediment and soils of the basin. Reduced loading through the TMDL is expected to help remediate these legacy nutrient effects. While the TMDL for these segments was developed in 2009, BMAPs, which include through legacy load management and removal (i.e., muck dredging and algal harvesting) were not adopted until 2013 (Banana River Lagoon Stakeholders and FDEP 2013, Central Indian River Lagoon Stakeholders and FDEP 2013, North Indian River Lagoon Stakeholders and FDEP 2013). Pending TMDL implementation, the presence of in-basin legacy nutrients taken with drought conditions in recent years may have enhanced nutrient flux/internal recycling of legacy nutrient loads within sediments and exchanged with the water column.

The BMAP annual reports for the Indian River Lagoon and St. Lucie basins verify that the efforts are increasingly reducing nutrient loading within the basin, with the greatest reductions occurring in the first two years of implementation (FDEP 2014b, c, d, e, 2015b, c, d, e, 2016b, c, d, e). The response of individual waterbodies to reductions in loadings is complicated by the presence of legacy loads in the sediment and soils of the basins and variability in hydrological discharge.

Table 7. Results of Regression Analyses of Post-Total Maximum Daily Load Changes in Nutrient Indicators for Those Data Meeting Restrictions for Simple Linear Regression.

Estuary Segment	Indicator	Slope	Standard Error	Significance
Banana River Lagoon	Chl-a , log TN, TN log Chl-a log TP TP	ns[a] 0.0629 0.0266 0.0012	ns 0.0156 0.0071 0.0004	ns 6.37E-05 <0.0002 0.0031
Central Indian River Lagoon	log Chl-a log TP TN	ns 0.0150 0.0112	ns 0.0044 0.0027	ns <0.0007 3.01E-05
North Indian River Lagoon	log TN, log TP log Chl-a	ns 0.0829	ns 0.01052	ns 6.14E-15
Lower St. Johns River and Tributaries (predominantly marine)	log TN	-0.0208	0.0048	1.60E-05
St. Lucie Estuary	Chl-a log TN	ns -0.0746	ns 0.0131	ns 3.03E-08
Upper North Fork St. Lucie River	log Chl-a, Chl-a	ns	ns	ns
Upper South Fork St. Lucie River	Chl-a, log TP, TP log TN TN	ns -0.0197 -0.0264	ns 0.0059 0.0080	ns 0.0011 0.0012

[a] ns = not significant

Other measures of TMDL success would include water clarity as measured by secchi depth (Figure 16) and DO regime indicating non-eutrophic conditions (i.e., absence of super saturation at mid-day, non-lethal pre-dawn DO levels). Current DO data are not the continuous monitoring required to provide a reliable representation of DO regime for these systems, but evaluation of mid-day DO saturation in summer can be used flag the presence of conditions favoring supersaturation (i.e., warm weather, bright sunlight are optimal conditions for photosynthesis).

Figure 16. A Secchi Disk Measure of Water Clarity is the Depth at Which a Disk is Visible when Submerged in Water.

To control for the effects of seasonal differences, the available DO saturation and water clarity data for the estuaries with load-based NNC were subsetted to include data only from the warmer months of June through September and mid-day hours of 10 am to 2 pm. The limited data available did not suggest a change in the frequency or severity of supersaturation over time. Water clarity from secchi depth data provide mixed results. Clarity appears to be improving in the Central Indian River Lagoon and the Upper North Fork of the St. Lucie River but declining in the Banana River Lagoon and the North Indian River Lagoon. However, BMAPs for these waters were only recently implemented and not expected to provide detectable improvements.

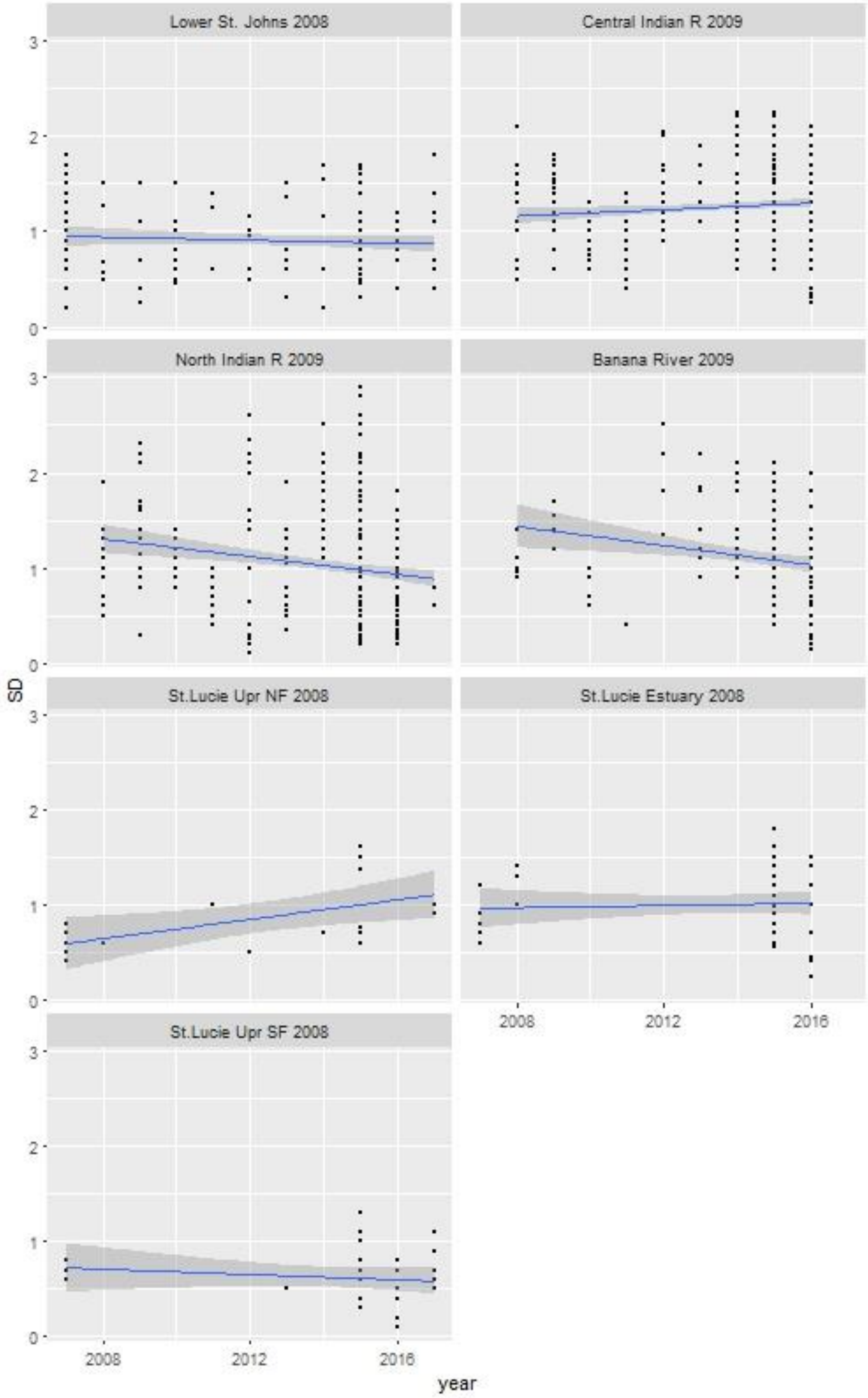

Figure 17. Post-Total Maximum Daily Load Observations of Secchi Disk Depth Visibility.

CONCLUSION: DO THE LOAD-BASED ESTUARY NUMERIC NUTRIENT CRITERIA PROMOTE OR SUSTAIN EUTROPHIC CONDITIONS?

Considering analysis results and information from BMAP documentation, taken with the knowledge that recovery from nutrient impairment takes many years, it is reasonable to conclude that the period of time covered by the available data is likely too short for nutrient management efforts to be detected. Individual waterbodies respond differently to changes in nutrient loading. The presence of legacy nutrients, external unmanaged sources, and hydrological variability can result in short term spikes or declines that are not directly attributable to managed loading reductions. Visual interpretation and regression analyses suggest that load-management is working towards recovering impaired systems. Ultimately, recovery from nutrient impairment requires the TMDL measures to first change the nutrient regime then achieve a somewhat steady-state condition that does not sustain eutrophication (Greening and Janicki 2006, Greening et al. 2011, Greening et al. 2014, Riemann et al. 2016, Staehr et al. 2017).

The NMFS concludes that the load-based NNC are expected to reverse, and not promote eutrophication because:

1) ***The data in hand for the Lower St Johns estuary segment indicate declining nutrient indicators in the years since the TMDL was implemented through a BMAP in 2008.***
2) ***Cases where nutrient indicators do not appear to have changed or appear to have increased since TMDL development may be explained by a gap in time before BMAP adoption (i.e., TMDLs had not yet been implemented through a BMAP until 2013 for Banana River Lagoon, Central Indian River Lagoon, North Indian River Lagoon, St. Lucie Estuary and the Upper South Fork of the St. Lucie River) and the interaction of hydrological conditions and legacy nutrient recycling that will continue to influence nutrient indicators in the water column (see Section 8.1.1) until TMDL implementation reduces external loading, stabilizes the system, and progresses towards TMDL goals.***
3) ***The BMAP annual reports for the Indian River Lagoon and St. Lucie basins verify that the efforts are increasingly reducing nutrient loading into these systems.***
4) ***Based on a meta-analysis of success factors for restoration of eutrophic lakes and estuaries (Gross and Hagy 2017), Florida's TMDL program is structured to achieve restoration goals for eutrophic systems.***

For these reasons, estuary segments with TMDL-based NNC will not be further addressed in the Response Analysis of this opinion.

8.2.2 Concentration-Based Numeric Nutrient Criteria

The remaining 27 estuary NNC for TN, TP and the nutrient response variable Chl-a, are concentration-based values derived from monitoring data representing reference or "healthy" (non-eutrophic) conditions or from mechanistic models predicting the fate and effects of anticipated pollutant loads. As discussed previously, relationships between nutrient concentrations and Chl-a are expected because elevated nutrient concentrations stimulate photosynthesis and can result in algal blooms. This relationship is the most prevalent in estuaries and enclosed water bodies where reduced water flushing allows for nutrient accumulation (Bricker et al. 2008).This section evaluates the concentration-based NNC to determine if they achieve the desired end, Chl-a at reference levels indicated by the Chl-a NNCs. The intent of the exposure analysis for segments with concentration-based NNC is not to identify impairment, calculate alternative criteria, or create a predictive model for Chl-a responses over the range of TN and TP. These analyses were conducted to determine whether achievement of concentration-based estuary NNC for TP and TN is consistently associated with Chl-a concentrations at or below the Chl-a NNC. The decision process for this approach is explained in greater detail below.

Most of these NNC are based on reference conditions using data from a period of time when the estuary segment itself was not considered to be impaired (i.e., reference period) or using data from an unimpaired estuary (i.e., reference ecosystem, or reference system) that is similar to the estuary segment of interest. The segment-specific NNC arrived at using these approaches range broadly (see Table 1).

The exposure analysis proceeds in two phases: first, using single sampling events within each estuary segment to evaluate the occurrence of Chl-a exceedences, and second, using aggregated sampling data to evaluate the detection of Chl-a impairments when the aggregated TN and TP are within their NNC. The latter is how the NNC will be implemented for each estuary segment, so this Opinion will sometimes refer to that approach to the data as the “as-implemented” approach. In the first phase, monitoring data is evaluated as single sample events to examine the frequency at which Chl-a exceeded the Chl-a NNC relative to compliance with TN and TP NNC in the same sample event. The contingency and conditional probability analyses used in this first phase are described in greater detail below. Environmental monitoring produces observational data, as opposed to experimental data, reflecting the many factors influencing biogeochemical cycles and biological responses at a discrete location at a specific time. These are snapshots in space and time and do not integrate the tidal, diurnal, and seasonal variation inherent in ecological systems.

The second phase evaluates the monitoring data as the NNC are implemented to integrate these factors. Florida implements most of the concentration-based NNC by aggregating sampling event data into open water, area wide annual geometric means (i.e., AGM) and comparing these with NNC. A segment is identified as impaired if more than one AGM NNC exceedence occurs in three consecutive years. For some estuaries, where data are sufficient, Florida evaluates the frequency of exceedences over seven consecutive years, identifying a segment as if more than 10 percent of observations exceed the NNC (i.e., PCT) over that period. In short, implementation of the NNC applies a specific frequency and sufficiency of data indicating a pattern of NNC exceedances to denote impairment, which indicates conditions favoring eutrophy. Because we are interested in the NNC are protective, and not whether the estuary segments are impaired, the phase two, “as implemented” analysis examines only those cases where the TN and TP were within their respective NNC to determine whether the implementation strategy identifies Chl-a exceedences or impairments over three-year time steps and how these exceedences relate to other years where the Chl-a AGM (or PCT) did not exceed the threshold.

THE DATA

Data used in deriving the criteria include TN, TP, and Chl-a data from the Impaired Surface Waters Rule (IWR) Run 49 database and additional datasets that were not in this IWR database, including data in the Florida STORET database. STORET data were combined with those available in IWR Run 49 to ensure that, where available, data records for each estuary extended through the end of 2013. The analyses used sampling events that were complete cases, that is, sampling events with TN, TP, and Chl-a data. Only those data flagged under EPA as “y” (i.e., acceptable data quality) were used. Data used by Florida in determining reference period standards were purposefully not used in the first phase of the analysis because those data would bias the analysis toward confirming the NNC as protective. For these reference period-based NNC, NMFS' analysis extracted data from 2014 through 2016 from the IWR Run 52 and 53 data along with recently collected data reported in the STORET database. For estuary segments with reference system-based standards, the data used extend to 1982. The estuary segments with NNC based on reference systems include Apalachicola Offshore, Indian River Lagoon between Hobe Sound and St. Lucie, Indian River Lagoon from Ft. Pierce Inlet to Indian River County Line, Loxahatchee River Estuary (Southwest Fork), Manatee Creek, Tomoka Basin, and Tomoka River Estuary.

APPROACH TO THE FIRST PHASE ANALYSES

Environmental monitoring data often do not meet the distributional requirements for analyses typically applied to formal controlled studies (e.g., linear regression) and our purpose is not to derive alternative NNC or predictive models, so the first phase analyses were approached categorically. The Chl-a monitoring data were classified as either belonging to sampling events where TP and TN concentrations were at or below their respective NNC (i.e., consistent with NNC) or belonging to events where TN and/or TP concentrations were elevated over their NNC (i.e., elevated nutrients). Sampling events where the TP and TN levels were consistent with NNC would be expected to have Chl-a levels at or below the Chl-a NNC (i.e., "perfect compliance"). Similarly, sampling events where the TP and TN levels were above the NNC would be expected to have elevated Chl-a levels at a higher frequency (i.e., more observations) or greater severity (i.e., observations exceeding the Chl-a NNC to a large degree). This requires the assumption that the Chl-a NNC are sufficiently representative of reference conditions. As stated previously, NMFS concludes, after review of supporting technical documents ((FDEP 2015a); (USEPA 2012b)) supporting the determination of estuary Chl-a NNC that it is reasonable to expect the Chl-a NNC to represent reference (i.e., non-eutrophic) conditions and may therefore be used as an indicator to evaluate whether the TN and TP NNC support eutrophic conditions.

This opinion's analysis applies conditional probability, expressed in contingency tables (see Table 8 for example), to describe the frequency of a Chl-a exceedence given TN and TP NNC-compliance.

Table 8. Example Contingency Table.

	Condition A	**Condition B**	
Response A	Frequency of Response A under Condition A	Frequency of Response A under Condition B	Frequency of Response A
Response B	Frequency of Response B under Condition A	Frequency of Response B under Condition B	Frequency of Response B
	Frequency of Condition A	Frequency of Condition B	

Fisher’s exact test of independence is a statistical method applied that can be applied to contingency tables to test a null hypothesis that the response is independent of the condition (e.g., Chl-a compliance with the Chl-a NNC is not influenced by whether or not the TN and TP are compliant with their respective NNCs). A significant result, indicated by a P value under 0.05, would lead to rejection of that null hypothesis and the conclusion that the data fail to demonstrate the response is independent of the condition. In other words, if the test result for is P,<0.05, Chl-a compliance with the Chl-a NNC appears to be influenced by whether the TN and TP concentrations are compliant with their NCC. This interpretation requires some qualification. When sample sizes are small or unevenly distributed, the power of such tests can be poor. Power is the probability that a test will correctly detect an effect that is present. For example, consider the case in Table 9. A majority of observations, 36 out of 46, are compliant with all NNC and there are no observations of elevated Chl-a among the few TN and/or TP>NNC data that are available. The P value for this contingency is >0.999, suggesting TN/TP compliance does not influence Chl-a compliance. These data appear to be from a segment that is consistently NNC-compliant and there is very little information on noncompliant cases, making it difficult to determine whether TN and TP compliance influences Chl-a. Calculating the power of a test takes into account both the magnitude of differences in Chl-a NNC-compliance between the two conditions, NNC-compliance and noncompliance, and the amount of information available to compare these conditions (i.e., the number of observations for each condition). The result of a power analysis is a probability that an effect can be detected with the data in hand. For example, a power of 0.75 would be interpreted as a 75 percent chance that the test will detect an effect if it is present. The power of a Fisher’s exact t test for the data in Table 9 is 0.

Nevertheless, a decision must still be made regarding NNC performance with the data in hand. Looking at these data, we would not necessarily conclude that TN and TP NNC-compliance in this segment does not influence Chl-a compliance because the majority of observations meet the TN and TP NNC, and, among these, the majority also meet the Chl-a NNC. It is reasonable to conclude, based on overall compliance, that these data provide evidence that achieving the TN and TP NNC results in Chl-a within the Chl-a NNC in 84 percent of cases (i.e. 36 out of 43 observations). This is the conditional probability of achieving the Chl-a NNC under the condition of TN and TP NNC-compliance.

Table 9. Example Contingency Table Where Fisher's Exact Test of Independence Results in a P value > 0.999 with a Power of 0 Percent Probability of Detecting an Effect when it is Present.

	TN/TP > NNC	TN/TP ≤ NNC	
Elevated Chl-a	0	7	7
Chl-a ≤ NNC	3	36	39
	3	43	

Contingency tables are also used in calculation of conditional probability based on Bayes theorem (Equation 1) which describes the probability of a response or result (e.g., elevated Chl-a), given knowledge of related conditions (e.g., TN and TP at NNC). In equation form:

Equation 1. Bayes Theorem.

$$P(A \mid B) = \frac{P(B \mid A)}{P(B)} * P(A)$$

$$P(\text{Elevated Chl-a} \mid \text{TN\&TP at NNC}) = \frac{P(\text{TN\&TP at NNC} \mid \text{Elevated Chl-a})}{P(\text{TN\&TP at NNC})} * P(\text{Elevated Chl-a})$$

Conditional probability is an approach to gathering evidence from available data to support a decision. Unlike the Fisher's exact test, there is no test statistic to provide a definitive threshold identifying cases whether the condition (i.e., TN-TP NNC-compliance) is influencing the response (i.e., Chl-a). Interpretation of failure rate is related to the gravity of the decision to be made and therefore relies to a certain degree on professional judgement regarding the plausibility of the relationship and consideration of data sufficiency. Accepting that the relationship is plausible and the data are sufficient (sensu, statistical power), contrast a 20 percent chance of traffic triggering a dam failure within the next 10 years versus a 20 percent chance of spilling your cup of coffee while multitasking tomorrow morning. The interpretation and decision is also based on the magnitude of the failure. A decision maker might ask: How much water (or tailings) could be released upon dam failure and who is in the path of the discharge? In the case of the coffee, consider how much coffee may be spilled, how hot it is, and where it may spill. In terms of the performance of the NNC: chronic, but very low-level Chl-a exceedences may be less concerning than severe episodic exceedences.

THE DECISION PROCESS

If monitoring data for an estuary segment indicate Chl-a NNC-compliance for all sampling events, or AGM, that were compliant with the TN and TP NNC, and some of the TN and TP concentrations were close to their respective NNC: this would provide clear evidence that the NNC are not expected to promote or sustain eutrophication. On the other hand, if Chl-a was consistently elevated in all sampling events, or AGM, that were compliant with the TN and TP NNC, this would provide clear evidence that the NNC are not protective. However, not all monitoring data are expected to provide such clear-cut evidence.

Decisions on whether Chl-a compliance is consistently associated with TN and TP NNC-compliance are made based on the strength of evidence from both phases of the analysis. In the first phase: Fisher's exact test, the power of those tests, and the conditional probability of Chl-a NNC-compliance given TN and TP NNC-compliance, all taking into consideration the usefulness of the data in making this determination. In the second phase of the analysis, we consider whether the TN and TP data during years with elevated Chl-a are similar to years where the Chl-a was not elevated and whether there are factors influencing or possibly driving Chl-a levels in addition to TN and TP. For example, cases where the Chl-a NNC is exceeded, but TN and TP levels are similar to levels in cases where the Chl-a was not exceeded, would suggest other factors, and not TN and TP, were influencing elevated Chl-a concentrations. Such factors would include changes in water salinity, water turnover rates, and the occurrence of bloom events (see Section 8.1.1). The degree of exceedences is also a factor. Are the observed exceedences relatively small (i.e., within 10 percent of the NNC) and likely to be an artifact of noise in the data? Are conditions, such as chronic bloom events, potentially increasing the amount of noise in the data? These considerations are integrated into the decision process applied below.

DO CONCENTRATION-BASED ESTUARY NNC PROMOTE OR PREVENT EUTROPHIC CONDITIONS?

Analyses of Sampling Events

Analysis of sampling event data uses the available data for each estuary segment to evaluate how well sampling events meeting the TP and TN NNC correspond with Chl-a levels meeting the Chl-a NNC. For reference period-based NNC, the analysis is restricted to data collected after 2013, because data used in the derivation of the standards would produce a self-confirmatory result. Otherwise, available data meeting applicable data quality standards were included in this analysis. However, only 18 of these segments have monitoring data available that were not used in the derivation of the NNC and included data for all three nutrient indicators, TN, TP, and Chl-a. In particular, dsampling event data for all three nutrient indicators, TN, TP, and Chl-a, were not available for five estuary segments: the Aucilla Offshore, Aucilla River Estuary, Ochlockonee River Estuary, and Palm Beach County ICWW, Ochlockonee/Alligator Harbor Offshore. The suitability of the NNC for these segments will have to be inferred from the performance of the NNC for other segments with NNC developed using the same methodology.

Estuary segment-specific NNC performance data are presented in Table 10 through Table 14. Sample size and the distribution of data among categories within these segments is important, as it influences power in the Fisher's exact test and the impact of a single event on the perceived proportion of Chl-a compliant or noncompliant events. For example, among small datasets, such as that for the Econfina River Estuary (Table 13), individual sampling events greatly influence any summary statistic, and can result in over-interpretation or misinterpretation. We see from the power statistic that a statistical test using these data have a 5 percent chance of detecting an effect of TN and TP NNC-compliance on Chl-a NNC-compliance. The following sections describe different scenarios estuary segment monitoring data fall into: segments with balanced data indicating positive evidence of TN and TP NNC-compliance resulting in Chl-a NNC-compliance and TN and TP NNC-noncompliance resulting in Chl-a NNC-noncompliance (Table 10), segments with abundant TN and TP NNC-noncompliant data providing evidence that elevated TN and TP is associated with elevated Chl-a (Table 11), segments with abundant TN and TP NNC-compliant data providing evidence that NNC-compliant TN and TP is associated with NNC-compliant Chl-a (Table 12), segments with sparse data providing no evidence for or against the protectiveness of the NNC (Table 13), and segments that do not provide data showing that TN and TP NNC-compliance influences Chl-a NNC-compliance (Table 4). Where Fisher's exact tests do not indicate an effect of TN and TP NNC-compliance on Chl-a NNC-compliance in all cases, this does not mean the data do not provide evidence for the evaluation of the NNC. For example, in unbalanced datasets when there is little data for TN and TP NNC-noncompliant cases, comparison of TN and TP NNC-compliant cases will have poor power and the Fisher's exact test result should be interpreted in context of t power of the test. The data still indicate the frequency at which Chl-a NNC-compliance occurs under TN and TP NNC-compliance.

Balanced data indicating positive evidence of TN and TP NNC-compliance resulting in Chl-a NNC-compliance and TN and TP NNC-noncompliance resulting in Chl-a NNC-noncompliance: For five segments, Alligator Harbor, Moorings Bay, Tomoka River Estuary, Indian River Lagoon from Ft. Pierce Inlet to Indian River County Line, and the ICWW between North Lake Worth Lagoon and Lower Loxahatchee River, the Fisher's exact test results appear to be driven by both Chl-a NNC-compliance under TN and TP NNC-compliance and by Chl-a NNC noncompliance among sampling events compliant with elevated TN and TP (Table 10). Meanwhile the Indian River Lagoon between Hobe Sound and St. Lucie segment results indicate potentially over protective NNC: while the Chl-a NNC target is met convincingly when TN and TP meet their NNC, Chl-a compliance is also more frequent than not (i.e., at 69 percent) even when the TN and TP exceed their NNC. The relationship between Chl-a and TN and TP levels is consistent with expectations in this segment, with the Chl-a NNC target being met twice as frequently whenthe TN and TP NNC are met.We are primarily interested in what happens under the condition of TN and TP NNC-compliance, and these data indicate that Chl-a levels within the NNC are achieved in a majority of sampling events that are compliant with the TN and TP NNC.

Table 10. Contingency Analyses Indicating Positive Evidence of an Influence of Total Nitrogen and Total Phosphorous Numeric Nutrient Criteria-compliance on Chlorophyll-a Numeric Nutrient Criteria-compliance.

	TP and/or TN Above NNC		TP and TN Within NNC			Fisher's Exact Test	Power (probability of detecting an effect in a statistical test)
Estuary Segment	Elevated Chl-a	Chl-a Within NNC	Elevated Chl-a	Chl-a Within NNC	Probability of Elevated Chl-a within TN TP NNC		
Alligator Harbor	13	1	3	11	21%	<0.001	0.98
Moorings Bay	57	19	23	99	19%	<0.001	>0.99
Tomoka River	65	92	52	124	30%	0.029	0.58
Indian River Lagoon: Ft. Pierce Inlet to Indian River County Line	135	82	37	102	27%	<0.001	>0.99
Indian River Lagoon between Hobe Sound and St. Lucie	73	163	24	136	15%	0.03	0.97
ICWW between North Lake Worth Lagoon and Lower Loxahatchee River	11	3	11	24	31%	0.004	0.83

The NMFS concludes that the data for six estuary segments (Alligator Harbor, Moorings Bay, Tomoka River Estuary, Indian River Lagoon from Ft. Pierce Inlet to Indian River County Line, Indian River Lagoon between Hobe Sound and St. Lucie, and the ICWW between North Lake Worth Lagoon and Lower Loxahatchee River) provide evidence that the NNC for TN and TP do not promote or sustain eutrophication because, when TN and TP are compliant with their respective NNC, Chl-a is consistently compliant with its NNC.

Abundant TN and TP NNC-noncompliant data providing evidence that elevated TN and TP is associated with elevated Chl-a: For Manatee Creek and the Mosquito River Lagoon segments from Edgewater to Oak Hill, from Oak Hill to the Southern Terminus, and from Ponce de Leon to Edgewater (Table 11), the Fisher's exact test results appear to be driven by a high frequency of Chl-a exceedences among sampling events with elevated nutrients (i.e., TN and/or TP concentrations were above their respective NNC).

Table 11. Contingency Analyses Indicating Association of Elevated Total Nitrogen and Total Phosphorous with Elevated Chlorophyll-a Numeric Nutrient Criteria.

	TP and/or TN Above NNC		TP and TN Within NNC				
Estuary Segment	**Elevated Chl-a**	**Chl-a Within NNC**	**Elevated Chl-a**	**Chl-a Within NNC**	**Conditional Probability of Elevated Chl-a within TN TP NNC**	**Fisher's Exact Test**	**Power (probability of detecting an effect)**
Manatee Creek	68	4	8	3	67%	0.046	0.51
Mosquito River Lagoon							
Edgewater to Oak Hill	83	16	9	8	53%	0.008	0.75
Oak Hill to Southern Terminus	122	13	10	13	43%	<0.001	>0.999
Ponce de Leon to Edgewater	32	12	4	8	33%	0.018	0.67

Given the relatively small number of sampling events that were compliant with the TN and TP NNC, each individual incident of elevated Chl-a strongly influences failure rate (i.e., percent elevated Chl-a among TN and TP NNC-compliant events). It is therefore not surprising that high failure rates are observed among these data.

> ***The NMFS concludes that data for four estuary segments (Manatee Creek and the Mosquito River Lagoon segments from Edgewater to Oak Hill, from Oak Hill to the Southern Terminus, and from Ponce de Leon to Edgewater) confirm the expectation that, when TN and TP exceed their respective NNC, Chl-a would exceed the Chl-a NNC. However, since there are so few observations where the TN and TP are compliant with their respective NNC, these data are not useful in determining whether or not the NNC for TN and TP promote or sustain eutrophication.***

Abundant TN and TP NNC-compliant data providing evidence that NNC-compliant TN and TP is associated with NNC-compliant Chl-a: The majority of sampling events for the Spring Warrior Offshore, and the Middle and Lower St. Marys River segments were compliant with the Chl-a, TN, and TP NNC (Table 12).

Table 12. Contingency Analysis for Estuary Segments Dominated by Sampling Events that were Compliant with the Total Nitrogen and Total Phosphorous Numeric Nutrient Criteria.

	TP and/or TN Above NNC		TP and TN Within NNC				
Estuary Segment	**Elevated Chl-a**	**Chl-a Within NNC**	**Elevated Chl-a**	**Chl-a Within NNC**	**Conditional Probability of Elevated Chl-a within TN TP NNC**	**Fisher's Exact Test**	**Power (probability of detecting an effect in a statistical test)**
Spring Warrior Offshore	0	3	0	62	0%	>0.999	0
Middle St. Marys River	0	3	7	36	16%	>0.999	0
Lower St. Marys River	0	6	1	46	2%	>0.999	0

This imbalance resulted in poor statistical power of comparisons. Rates of Chl-a compliance among TN and TP NNC-compliant sampling events for Spring Warrior Offshore, Middle and Lower St. Marys River were 95 percent, 78 percent, and 87 percent respectively.

The NMFS concludes that data for the Spring Warrior Offshore and the Middle and Lower St. Marys River estuary provide evidence that the NNC for TN and TP do not promote or sustain eutrophication because, when TN and TP are compliant with their respective NNC, Chl-a is consistently compliant with its NNC.

Sparse data providing no evidence for or against the protectiveness of the NNC: The Apalachicola Offshore, Econfina River Estuary, and St. Marks Offshore segments had data for fewer than 20 sampling events (Table 13). While Chl-a NNC-compliance was most frequent among TN and TP NNC-compliant sampling events for each segment, the extremely low power of such comparisons prevents use of these data.

Table 13. Contingency Analysis of Estuary Segments with Sparse Data.

	TP and/or TN Above NNC		TP and TN Within NNC				
Estuary Segment	**Elevated Chl-a**	**Chl-a Within NNC**	**Elevated Chl-a**	**Chl-a Within NNC**	**Conditional Probability of Elevated Chl-a within TN TP NNC**	**Fisher's Exact Test**	**Power (probability of detecting an effect in a statistical test)**
Apalachicola Offshore	0	1	0	3	0%	>0.999	0
Econfina River Estuary	1	1	3	9	25%	0.50	0.05
St. Marks Offshore	1	3	0	3	0%	>0.999	0.005

The NMFS concludes that data used in the phase one analysis for the Apalachicola Offshore, Econfina River Estuary, and St. Marks Offshore estuary segments are

too sparse to support a conclusion on whether or not compliance with the TN and TP NNC is consistently associated with compliance with the Chl-a NNC. (NOTE: post 2013 data were used for reference period-based NNC in the phase one analysis, all years of data were used for the phase two, "as implemented." analysis).

Data suggesting that NNC for TN and TP promote or sustain eutrophication: Fisher's exact tests for the Steinhatchee River Estuary, Pellicer Creek Estuary, and the Southwest Fork of the Loxahatchee River Estuary segments did not indicate a consistent association of Chl-a NNC-compliance with TN and TP NNC-compliance. Data for all three segments are characterized by low power. Data for the Steinhatchee and Pellicer Creek Estuary segments indicated that around half of the TN and TP NNC-compliant sampling events had elevated Chl-a. The relatively small sample sizes and poor power for these segments leads NMFS to conclude that the data are inadequate for evaluating the NNC. Meanwhile abundant data for the Southwest Fork of the Loxahatchee River Estuary indicate a high frequency of TN and TP compliance, but also a high frequency of elevated Chl-a among these TN and TP NNC-compliant sampling events (245 out of 319, or 77 percent). In this case, compliance with the TN and TP NNC appears to promote Chl-a levels above the Chl-a NNC.

Table 14. Contingency Analysis of Estuary Segments Indicating No Evidence of Total Nitrogen and Total Phosphorous Numeric Nutrient Criteria-compliance Influencing Chlorophyll-a Numeric Nutrient Criteria-compliance.

	TP and/or TN Above NNC		TP and TN Within NNC			Fisher's Exact Test	Power (probability of detecting an effect in a statistical test)
Estuary Segment	Elevated Chl-a	Chl-a Within NNC	Elevated Chl-a	Chl-a Within NNC	Probability of Elevated Chl-a within TN TP NNC		
Steinhatchee River	5	7	13	11	54%	0.725	0.08
Pellicer Creek	7	3	8	8	50%	0.428	0.11
SW fork Loxahatchee River Estuary	75	15	245	74	77%	0.197	0.22

The NMFS concludes that the TN and TP NNC for the Southwest Fork of the Loxahatchee appear to promote eutrophication based on abundant data indicating Chl-a levels elevated over the Chl-a NNC when TN and TP are within their respective NNC.

The NMFS also concludes that data available for the phase one analysis for the Steinhatchee River and Pellicer Creek estuary segments are too sparse to provide evidence on whether or not compliance with the TN and TP NNC would promote or sustain eutrophication. (NOTE: post 2013 data were used for reference period-basedf NNC in the phase one analysis, all years of data were used for the phase two, "as implemented," analysis).

Performance “As Implemented” Under Perfect TN TP NNC-compliance

The “as implemented” analysis evaluates TN and TP compliant data collected between 1982 through 2017 data for all estuary segments. The analysis provides an overall assessment of whether Chl-a concentrations meet their respective NNC when TN and TP compliant sampling events are aggregated to identify impairments as the NNC are intended to be implemented. Since the purpose of the analysis is to determine if Chl-a will meet its NNC where TN and TP meet their respective NNC, only sampling events where the TN and TP NNC are met (i.e., “perfect compliance”) are used. For review, segments with criteria expressed as AGM values shall not be exceeded more than once in a three-year period, regardless of the variance among data within years. For segments with criteria expressed in terms of percentiles (i.e., PCT), the criteria shall not be exceeded in more than 10 percent of the measurements over the most recent consecutive seven-year period. For criteria expressed as long-term averages (LTAVG), the long-term average shall be based on data from the most recent seven-year period and shall not be exceeded. Since the data are truncated to include only those sampling events with TP and TP NNC-compliance, this evaluation treats the AGM calculated from these data as though all other requirements, such as the representation of each season, are met.

Table 15. Summary of Concentration-based Numeric Nutrient Criteria Implementation Frameworks.

Segment	Chl-a Target	TN Criteria	TP Criteria	Method
Apalachicola Bay and Alligator Harbor				
Alligator Harbor	6.0 µg/L (AGM)	0.42 mg/L (AGM)	0.028 mg/L (AGM)	ref. period
Apalachicola Offshore	8.2 µg/L (PCT)	0.57 mg/L (PCT)	0.032 mg/L (PCT)	ref. system
Big Bend and Apalachee Bay				
Aucilla Offshore	9.5 µg/L (PCT)	0.60 mg/L (PCT)	0.025 mg/L (PCT)	ref. period
Aucilla River Estuary	2.2 µg/L (PCT)	0.89 mg/L (PCT)	0.080 mg/L (PCT)	
Econfina River Estuary	4.9 µg/L (AGM)	1.14 mg/L (AGM)	0.101 mg/L (AGM)	model
Ochlockonee River Estuary	9.2 µg/L (PCT)	0.86 mg/L (PCT)	0.067 mg/L (PCT)	ref. period
Ochlockonee/Alligator Harbor	8.2 µg/L (PCT)	0.57 mg/L (PCT)	0.032 mg/L (PCT)	
Spring Warrior Offshore	8.3 µg/L (PCT)	0.67 mg/L (PCT)	0.047 mg/L (PCT)	
St. Marks Offshore	8.0 µg/L (PCT)	0.63 mg/L (PCT)	0.045 mg/L (PCT)	
St. Marks River Estuary	6.0 µg/L (PCT)	0.70 mg/L (PCT)	0.044 mg/L (PCT)	
Steinhatchee River Estuary	3.9 µg/L (AGM)	0.86 mg/L (AGM)	0.062 mg/L (AGM)	
Charlotte Harbor/Estero Bay				
Moorings Bay	8.1 µg/L (AGM)	0.85 mg/L (PCT)	0.040 mg/L (PCT)	ref. period
Guana River/Tolomato River/Matanzas River (GTM) Estuary				
Pellicer Creek Estuary	4.3 µg/L (AGM)	1.10 mg/L (AGM)	0.123 mg/L (AGM)	ref. period
Halifax and Tomoka River Estuaries				
Tomoka River Estuary	7.2 µg/L (AGM)	1.24 mg/L (AGM)	0.132 mg/L (AGM)	ref. system
Indian River Lagoon System, including Banana River Lagoon and Mosquito River Lagoon				
Hobe Sound to St. Lucie	6.9 µg/L (PCT)	0.63 mg/L (AGM)	0.060 mg/L (AGM)	ref. system
Ft. Pierce to Indian River County Line	4.7 µg/L (AGM)	0.72 mg/L (AGM)	0.070 mg/L (AGM)	ref. period
St. Lucie to Ft. Pierce	4.7 µg/L (AGM)	0.72 mg/L (AGM)	0.070 mg/L (AGM)	
Intracoastal Waterway (ICWW)				
Palm Beach County ICWW	13.4 µg/L (PCT)	1.17 mg/L (PCT)	0.146 mg/L (PCT)	ref. period
North Lake Worth Lagoon to Lower Loxahatchee River	4.7 µg/L (AGM)	0.66 mg/L (AGM)	0.035 mg/L (AGM)	
Loxahatchee River Estuary				
SW Fork Loxahatchee River	5.5 µg/L (AGM)	1.26 mg/L (AGM)	0.075 mg/L (AGM)	ref. system
Mosquito River Lagoon				
Edgewater to Oak Hill	3.4 µg/L (AGM)	0.65 mg/L (AGM)	0.048 mg/L (AGM)	ref. period
Oak Hill to Southern Terminus	2.5 µg/L (AGM)	1.14 mg/L (AGM)	0.034 mg/L (AGM)	
Ponce de Leon to Edgewater	4.0 µg/L (AGM)	0.51 mg/L (AGM)	0.049 mg/L (AGM)	
St. Lucie Estuary				
Manatee Creek	5.9 µg/L (AGM)	0.72 mg/L (LTAVG)	0.081 mg/L (LTAVG)	ref. system
St. Marys River				
Lower St. Marys River	12.9 µg/L (AGM)	0.77 mg/L (AGM)	0.181 mg/L (AGM)	ref. period
Middle St. Marys River	8.0 µg/L (PCT)	1.12 mg/L (AGM)	0.113 mg/L (AGM)	
Upper St. Marys River	3.0 µg/L (PCT)	1.35 mg/L (AGM)	0.093 mg/L (PCT)	

At least three consecutive years of data are required to evaluate segments with AGM-implemented NNC and at least seven consecutive years of data are required for NNC implemented as PCT or LTAVG. Perfect TN and TP compliance data were available for the necessary number of years for the estuary segments identified in Table 16.

Table 16. Estuary Segments with Sufficient Total Nitrogen and Total Phosphorous-Compliant Data for the

“As Implemented” Analysis.

Estuary Segment	Implementation method for Chl-a	Years of TN and TP-compliant Data
Big Bend and Apalachee Bay		
Steinhatchee River Estuary	AGM	5
Charlotte Harbor/Estero Bay		
Moorings Bay	AGM	10
Guana River/Tolomato River/Matanzas River (GTM) Estuary		
Pellicer Creek Estuary	AGM	31
Halifax and Tomoka River Estuaries		
Tomoka River Estuary	AGM	30
Indian River Lagoon System, including Banana River and Mosquito River Lagoons		
between Hobe Sound and St. Lucie	PCT	16
from Ft. Pierce Inlet to Indian River County Line	AGM	21
from St. Lucie Estuary to Ft. Pierce Inlet	AGM	19
Intracoastal Waterway (ICWW)		
ICWW between North Lake Worth Lagoon and Lower Loxahatchee River	AGM	12
Loxahatchee River Estuary		
Loxahatchee River Estuary (Southwest Fork)	AGM	12
Mosquito River Lagoon		
Edgewater to Oak Hill	AGM	25
Oak Hill to the Southern Terminus	AGM	28
Ponce de Leon to Edgewater	AGM	26
St. Marys River Estuary		
Upper St. Marys River	AGM	9

The AGM-implemented NNC: Exceed NNC No More than Once in Three Consecutive Years

Only the Southwest Fork of the Loxahatchee and Moorings Bay had data consistently indicating Chl-a NNC-compliance (i.e., Moorings Bay) or NNC-noncompliance (i.e., Southwest Fork of the Loxahatchee) in all years. Most monitoring data for the AGM-implemented Chl-a NNC include data for segments where the TN and TP data are NNC compliant, but Chl-a is elevated in one or more years. Contributions of other factors enhancing Chl-a are suggested when TN and TP levels are similar among years, but Chl-a NNC is exceeded in only some of those years. Non-protectiveness of the NNC is indicated when the TN and TP AGM are greater in years where the Chl-a AGM exceeds the Chl-a NNC. For example, contrast the distribution of TN and TP NNC-compliant AGM among Chl-a NNC-compliant and noncompliant data for Pellicer Creek and the Steinhatchee River (Table 17). The TN data for Pellicer Creek, when the Chl-a AGM was within the NNC, averaged 0.68 mg TN/L with a standard deviation (sd) +/-0.14 mg/L. This gives a range of 0.54-0.82 mg TN/L under Chl-a NNC-compliance. The Pellicer Creek TN AGM in years where the Chl-a AGM exceeded the NNC averaged 0.54 mg TN/L with an sd of +/-0.26 mg/L, giving a range of 0.28-0.8 mg TN/L. The two ranges overlap and the TN levels can be considered similar under Chl-a NNC-compliance and noncompliance. Meanwhile for the Steinhatchee River, the TN ranges observed under Chl-a NNC-compliance and noncompliance do not overlap: 0.23 to 0.39 mg TN/L versus 0.67 to 0.87 mg TN/L. In this case, it appears that TN and TP AGM compliant with the NNC, but at higher concentrations, result in elevated Chl-a.

Table 17. Comparison of the Total Nitrogen and Total Phosphorous Observations for Two Estuary Segments.

Chl-a Status	Pellicer Creek		Steinhatchee River	
	TN mg/L +/-sd	TP mg/L +/-sd	TN mg/L +/-sd	TP mg/L +/-sd
Chl-a within NNC	0.68+/-0.14	0.08+/-0.01	0.31+/-0.08	0.02+/-0.002
Chl-a above NNC	0.54+/-0.26	0.08+/-0.02	0.75+/-0.12	0.04+/-0.01

In addition, in cases where the Chl-a NNC are frequently exceeded under TN and TP NNC-compliant AGM, the history of nutrient conditions within the segment are taken into consideration, as the occurrence of chronic blooms (e.g., due to temporal lags and spatial variability in nutrient stimulation – bloom initiation and algal nutrient consumption, see Section 8.1.1), changes in benthic communities, and drops in salinity can influence Chl-a.

The analyses for the NNC terms of the AGM are discussed in detail below.

Estuary Segment with Elevated Chl-a Occurring with TN and TP NNC in All Years: As suggested in the phase one contingency analysis, among AGM calculated using only TN and TP NNC-compliant sampling events, elevated Chl-a occurred frequently within the Southwest Fork of the Loxahatchee River Estuary. In fact, all Chl-a AGM exceeded the NNC over 12 years (Figure 18).

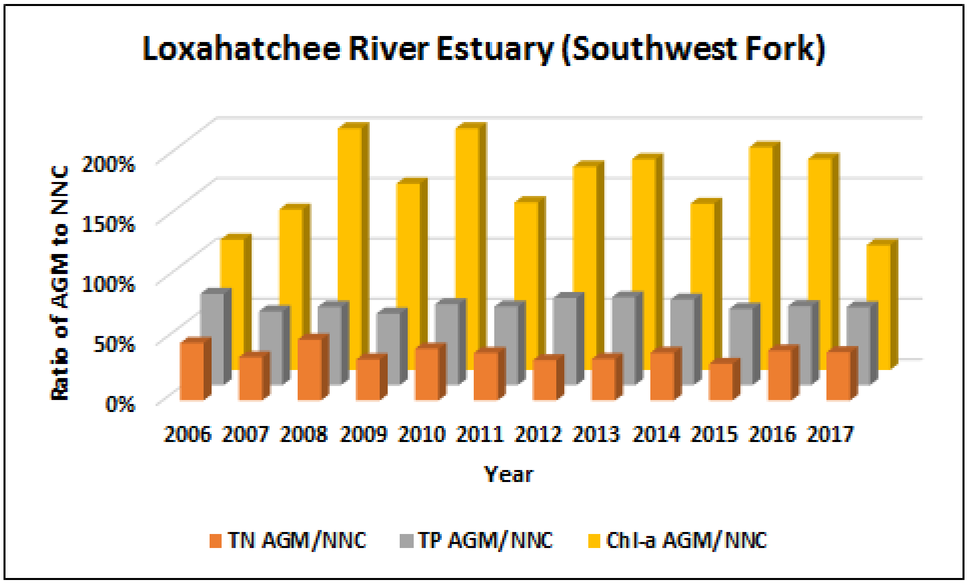

Figure 18. Annual Geometric Means for the Southwest Fork of the Loxahatchee River Estuary Showing Consistently Elevated Chlorophyll-a under Total Nitrogen and Total Phosphorous Numeric Nutrient Criteria-Compliant Annual Geometric Means over 12 years.

According to the 2016 Loxahatchee River Water Quality Monitoring report (Stoner et al. 2016), the Southwest Fork of the Loxahatchee has been an area of concern over the last decade. The report suggests that the chronic exceedence of the Chl-a NNC threshold of 5.5 ug/L is likely due to stagnant water accumulating during closures of the S-46 flood control structure, which diverts excess freshwater runoff from the Northwest Fork to the Southwest Fork of the river. Stoner et al. (2016) conducted pair-wise comparisons between the freshwater station immediately upstream and the brackish station immediately downstream of the flood control structure. The Chl-a concentrations were over 41% higher at the brackish station when compared to the freshwater station. In addition, TP was 37% higher at the brackish station, but TN concentrations were found almost 43%. Stoner et al. (2016) also determined that Chl-a concentrations were over 137% higher in the brackish water station when 7 day flows were < 200 cfs (Mean ± SD = 2.5 ± 5.1) compared to when 7 day flows were above that rate (i.e., higher flushing). Positive correlations between Chl-a concentrations with TN and TP concentrations were reported, but this work did not examine whether elevated Chl-a occurred at NNC-compliant concentrations of TN and TP. The report also described a September 28, 2015 observation of and algae bloom in the Southwest Fork, just upstream of the brackish water station. A sample from this bloom was subsequently found to contain the toxic algae *Microcystis aeruginosa*. When aggregated as AGM, the data provide no evidence that Chl-a NNC can be achieved at the TN and TP NNC, This suggests the TN and TP NNC need to be revised to match the vulnerability of the system or conditions in the segment must be changed to reduce its vulnerabilty.

> ***The NMFS concludes that the TN and TP NNC for the Southwest Fork of the Loxahatchee River Estuary are expected to promote eutrophication because nutrient concentrations within the NNC are consistently associated with elevated***

Chl-a and the flood control structure was shown to create conditions that make this segment vulnerable to nutrient accumulation (Stoner et al. 2016).

Estuary Segment with NNC-compliant Chl-a Occurring with TN and TP NNC in All Years: Among AGM calculated using only TN and TP NNC-compliant sampling events, elevated Chl-a did not occur in any year among data for Moorings Bay. This suggests the TN and TP NNC would not promote or sustain eutrophication.

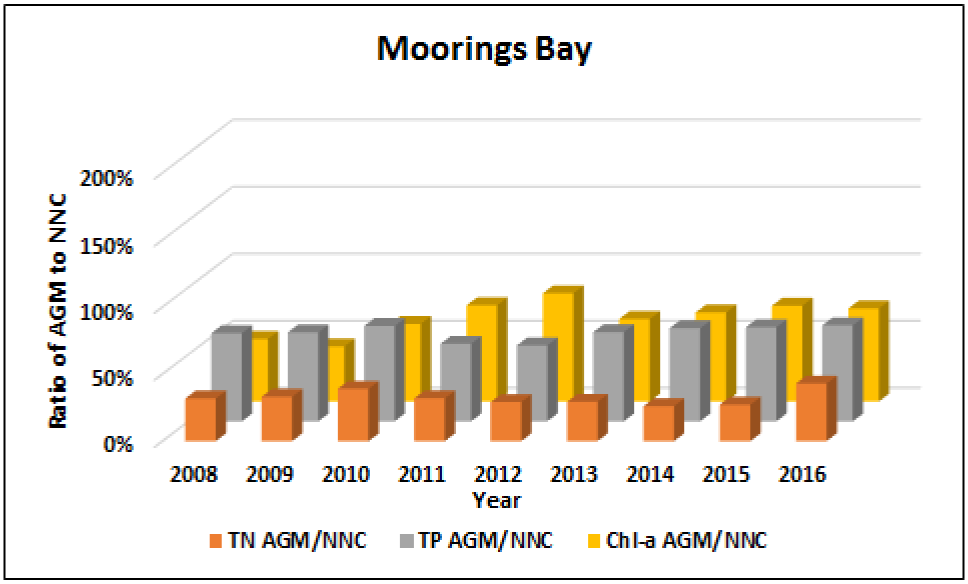

Figure 19. Annual Geometric Means for the Moorings Bay Showing Consistently Elevated Chlorophyll-a under Total Nitrogen and Total Phosphorous Numeric Nutrient Criteria-Compliant Annual Geometric Means over Nine years.

The NMFS concludes that the TN and TP NNC for Moorings Bay are not expected to promote eutrophication because nutrient concentrations within the NNC were associated with NNC-compliant Chl-a over nine years.

Estuary Segments with Elevated Chl-a Occurring with NNC-Compliant TN and TP AGM: Among AGM calculated using only TN and TP NNC-compliant sampling events, elevated Chl-a AGM occurred in one or more years for the ICWW segment between North Lake Worth Lagoon and Lower Loxahatchee River, two segments of the Indian River Lagoon Basin (from the St. Lucie Estuary to Ft. Pierce Inlet and from Ft. Pierce Inlet to Indian River County Line), three segments of the Mosquito Lagoon basin (from Ponce de Leon to Edgewater, from Edgewater to Oak Hill, and from Oak Hill to the Southern Terminus), the Pellicer Creek estuary segment and Tomoka River estuary segment. The sequence of elevated Chl-a among these data did not always result in identification of an impairment, as implemented by Florida, which requires two or more exceedences in three consecutive years. Occurrences of a Chl-a AGM above the Chl-a NNC are discussed to explore apparent inconsistencies and address bloom events.

Among AGM calculated using only TN and TP NNC-compliant sampling events in the ICWW between North Lake Worth Lagoon and Lower Loxahatchee River, impairment by elevated Chl-a was indicated for 2008-2010. However, the TN and TP AGMs in years where the Chl-a AGM was elevated did not differ substantially from years when the Chl-a AGM was within the Chl-a NNC (Table 18, Figure 20). This suggests interacting factors promoting Chl-a were present in the segment. This also suggests that TN and TP, when at levels within their respective NNC and in the absence of such factors, would not be expected to promote or sustain eutrophication (see Section 8.1.1 for discussion).

Table 18. Comparison of Intercoastal Water Way between North Lake Worth Lagoon and Lower Loxahatchee River Average Total Nitrogen and Total Phosphorous Annual Geometric Mean where the Chlorophyll-a Annual Geometric Mean was within, Versus Above, the Chlorophyll-a Numeric Nutrient Criteria.

	TN mg/L +/-sd	TP mg/L +/-sd
Chl-a within NNC	0.25+/- 0.03	0.02+/- 0.002
Chl-a above NNC	0.28+/- 0.05	0.02+/- 0.001

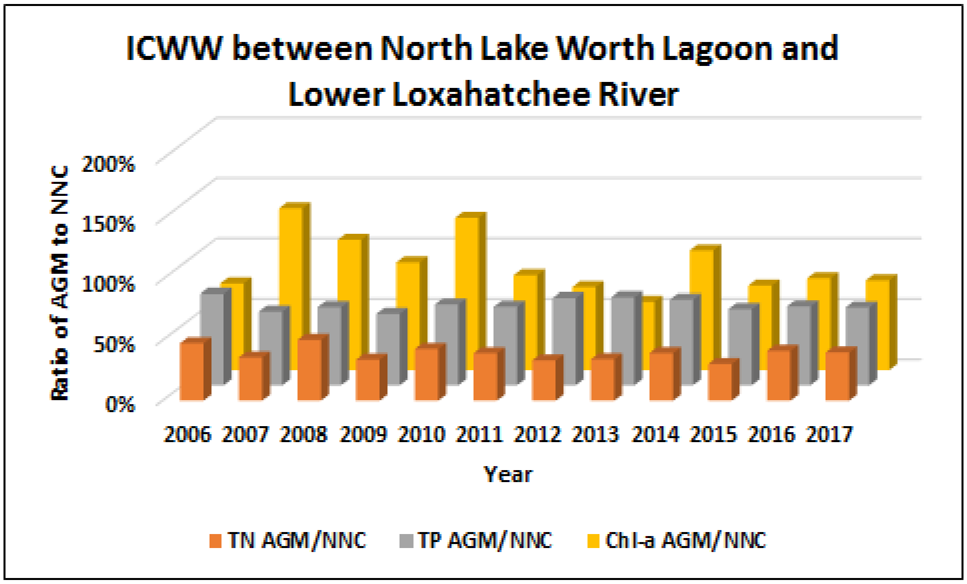

Figure 20. Annual Geometric Means for Intercoastal Water Way between North Lake Worth Lagoon and Lower Loxahatchee River Showing Elevated Chlorophyll-a with Total Nitrogen and Total Phosphorous Annual Geometric Means within the Numeric Nutrient Criteria Three Times Over 12 years.

The NMFS concludes that the TN and TP NNC for the ICWW between North Lake Worth Lagoon and Lower Loxahatchee River are not expected to promote eutrophication because nutrient concentrations within the NNC are associated with NNC-compliant Chl-a in most cases, and where spikes in Chl-a occur, the TN and TP AGM are within one standard deviation of the TN and TP AGM when the Chl-a AGM was within the NNC.

There is no indication of Chl-a impairment among AGM calculated using only TN and TP NNC-compliant sampling events in the Indian River Lagoon segment from St. Lucie Estuary to Ft. Pierce Inlet. However, the Chl-a AGM was about twice the Chl-a NNC in 1990 and four times the NNC 2013 (Figure 21). The elevated Chl-a in 2013 was likely associated with the latter of the brown tide events occurring in 2012 and 2013 in the basin(NOAA/NCCOS 2013). In both instances the TN AGM, but not the TP AGM was higher, at greater than 90 percent of the TN NNC than in years where the Chl-a AGM was not elevated (Table 19).

Table 19. Comparison of Indian River Lagoon segments from St. Lucie Estuary to Ft. Pierce Inlet Average Total Nitrogen and Total Phosphorous Annual Geometric Mean where the Chlorophyll-a Annual Geometric Mean was within, Versus Above, the Chlorophyll-a Numeric Nutrient Criteria.

	TN mg/L +/-sd	TP mg/L +/-sd
Chl-a within NNC	0.47+/-0.12	0.04+/-0.01
Chl-a above NNC	0.67+/-0.01	0.04+/-0.01

The relevance of the TN AGM approaching, but not exceeding the TN NNC in 2013 is confounded by the expected disruption in nutrient-Chl-a dynamics due to the impact of the 2012 and 2013 brown tides. Table 20 indicates that, in this year, the Chl-a levels among TN and TP NNC-compliant sampling events were nearly identical to Chl-a levels among TN and TP NNC-noncompliant sampling events, providing evidence that other factors, and not the TN AGM influenced the Chl-a AGM. The NMFS looked for, but did not find information on algal bloom conditions within the basin in 1990. However the basin was known to be impaired by discharges from sewage plants and, in 1990, the Florida Legislature passed the Indian River Lagoon Act requiring most sewer plants to stop discharging into the lagoon (Indian River Lagoon National Estuary Program 2008). It is reasonable to expect that the elevated Chl-a AGM in 1990 among TN and TP NNC-compliant sampling events is also associated with additional conditions contributing to Chl-a (see Section 8.1.1 for a discussion of nutrient dynamics and factors influencing Chl-a).

Table 20. Comparison of Indian River Lagoon from St. Lucie Estuary to Ft. Pierce Inlet Total Nitrogen and Total Phosphorous in Years Where the Chlorophyll-a Annual Geometric Means Exceeded the Chlorophyl a Numeric Nutrient Criteria.

Nutrient Indicator	1990		2013	
	TN and TP above NNC	TN and TP within NNC	TN and TP above NNC	TN and TP within NNC
Chl-a µg/L	18.67+/-10.98	13.26+/-6.74	21.79+/-5.51	21.5+/-5.53
TN mg/L	0.73+/-0.31	0.51+/-0.16	0.82+/-0.08	0.67+/-0.04
TP mg/L	0.09+/-0.03	0.05+/-0.02	0.04+/-0.01	0.03+/-0

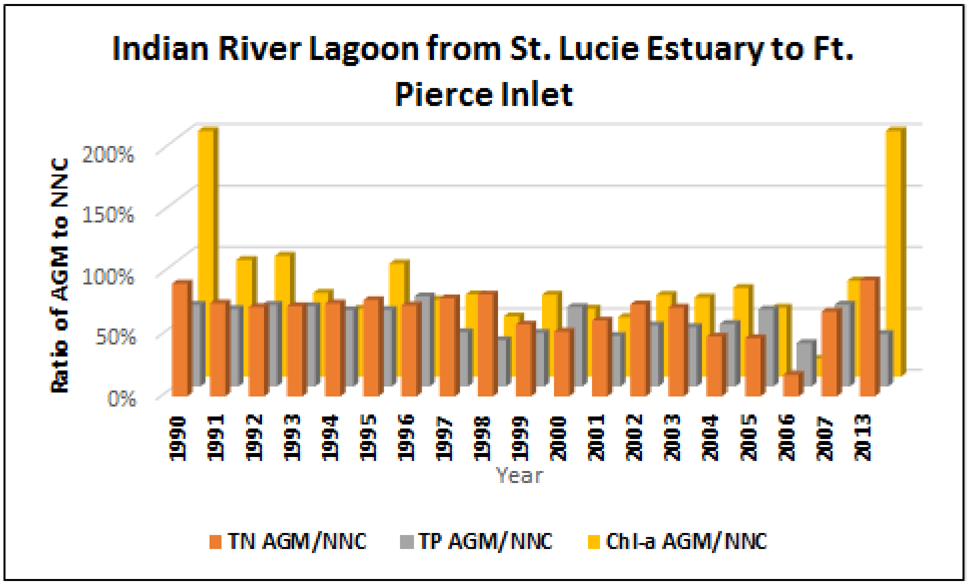

Figure 21. Annual Geometric Means for Indian River Lagoon from St. Lucie to Ft. Pierce Showing Elevated Chlorophyll-a with total Nitrogen and Total Phosphorous Annual Geometric Means within the Numeric Nutrient Criteria Twice over 19 Years.

The NMFS concludes that the TN and TP NNC for the Indian River Lagoon segment from St. Lucie to Ft. Pierce are not expected to promote eutrophication because, with the exception of an algal bloom-associated Chl-a spike in 2013 and likely Chl-a promoting conditions other than TN and TP in 1990, NNC-compliant Chl-a is consistently associated with TN and TP AGM that are compliant with their respective NNC.

Among AGM calculated using only TN and TP NNC-compliant sampling events in the Indian River Lagoon segment from Ft. Pierce Inlet to Indian River County Line, impairment by elevated Chl-a occurred in 1992 and 1993. During this period, the TN and TP AGM were 72-80 percent of their respective NNC (Figure 22). The TP AGM for Chl-a NNC-compliant years did not differ substantially from years where the Chl-a AGM was elevated (Table 21). While the average of the TN AGMs in years with elevated Chl-a was slightly higher than the average of years where the Chl-a AGM was not elevated, Chl-a was not elevated in individual years when the TN AGM exceeded the 1992 and 1993 TN AGM (see Figure 22, years 1994, 1997-98, 2002-03).

Table 21. Comparison of Indian River Lagoon from Ft. Pierce Inlet to Indian River County Line Total Nitrogen and Total Phosphorous Annual Geometric Means where the Chlorophyll-a Geometric Means were Within, Versus Above the Chlorophyll-a Numeric Nutrient Criterion.

	TN mg/L +/-sd	TP mg/L +/-sd
Chl-a within NNC	0.49+/-0.12	0.04+/-0.01
Chl-a above NNC	0.52+/-0.07	0.05+/-0.01

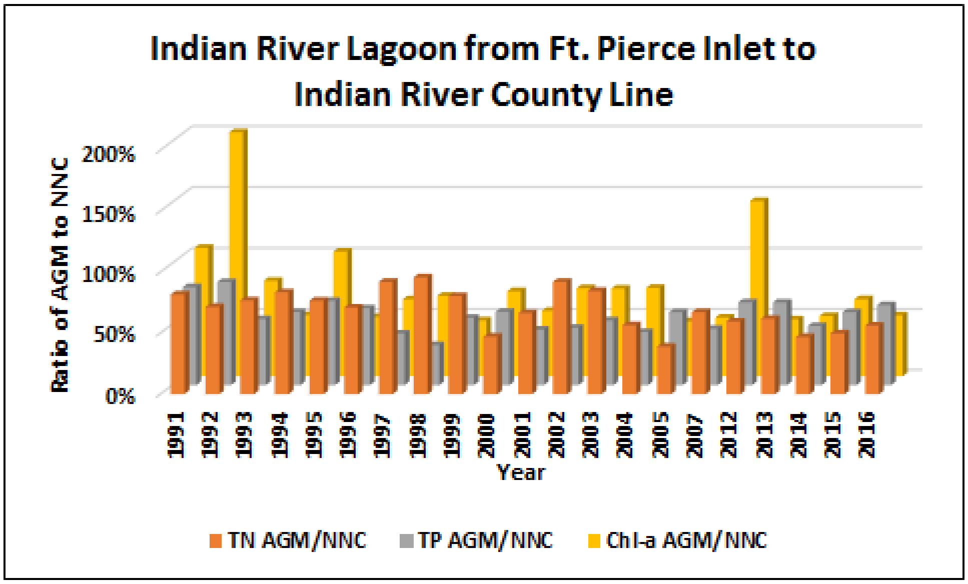

Figure 22. Annual Geometric Means for Indian River Lagoon from Ft. Pierce Inlet to the Indian River County Line Showing Elevated Chlorophyll-a with Total Nitrogen and Total Phosphorous Annual Geometric Means within the NTwice in 21 Years.

Among 21 years of data, the TN AGM, but not the TP AGM, also exceeded 80 percent of their respective NNC five times, yet did not result in an elevated Chl-a AGM. While this could suggest that the TP drove the Chl-a elevations in 1991 and 1992, a spike in Chl-a in 2012 suggests that TN and TP AGM levels did not drive the Chl-a AGM in that year. In 2012, 2013, and 2016, the TN and TP AGM were nearly identical, at around 60 and 70 percent of the TN and TP NNC, respectively, yet Chl-a was elevated only in 2012. Observations of similar TN and TP conditions not associated with elevated with Chl-a suggests that other factors, most likely inclusion of sampling events from the Indian River Lagoon 2012 brown tide event (NOAA/NCCOS 2013) in the AGM, contributes to the 2012 Chl-a AGM exceedence.

> ***The NMFS concludes that the TN and TP NNC for the Indian River Lagoon segment Ft. Pierce Inlet to the Indian River County Line are not expected to promote eutrophication because, with the exception of an algal bloom-associated Chl-a spike in 2012 and likely Chl-a promoting conditions other than TN and TP in the early 1990s, NNC-compliant Chl-a is consistently associated with TN and TP AGM that are compliant with their respective NNC.***

All three of the Mosquito River Lagoon segments had elevated Chl-a at the TN and TP NNC in recent years Among AGM calculated using only TN and TP NNC-compliant sampling events in (Figure 23). The Ponce de Leon to Edgewater and Edgewater to Oak Hill segments both had TN and TP AGM at or close to the NNC in 2015. The TN and TP-compliant data for the segment from Oak Hill to the Southern Terminus was flagged in this analysis as Chl-a impaired since 2012. However, the TN and TP AGMs in these years were not remarkably different from previous years (Table 22). The same is true for the Edgewater to Oak Hill and Oak Hill to Southern Terminus segments (Figure 23, Table 22). The association of elevated Chl-a in these three closely related segments over the same time period suggests conditions had changed to enhance water column Chl-a. Beginning in March of 2011, Mosquito River Lagoon and the neighboring Banana River Lagoon and Central Indian River Lagoon experienced a series of algal blooms with Chl-a concentrations exceeding 100 ug/L for nearly 8 months (St. Johns River Water Management District 2012). A brown tide bloom followed in 2012, reaching nearly 200 ug/L Chl-a and brown tides continue to occur in these waters (St. Johns River Water Management District 2017). A two-year drought and record cold winter preceeded the bloom. These conditions are suspected of increasing salinity and reducing populations of benthic organisms that graze on algae (Phlips et al. 2015).

Among the 27 years of data for these waters, Chl-a AGM met the Chl-a NNC more often than not when TN and TP AGM were compliant with their respective NNC. Under TN and TP NNC-compliance, the Chl-a NNC was exceeded a total of five times in the Ponce de Leon to Edgewater segment, three times in the Edgewater to Oak Hill segment and ten times in the Oak Hill to the Southern Terminus segment.

Table 22. Comparison of Mosquito River Lagoon Estuary Segment Average Total Nitrogen and Total Phosphorous Annual Geometric Means where the Chlorophyll-a Geometric Means were Within, Versus Above the Chlorophyll-a Numeric Nutrient Criterion.

Mosquito River Lagoon Segment	TN mg/L +/-sd at Chl-a NNC Status		TP mg/L +/-sd at Chl-a NNC Status	
	within NNC	above NNC	within NNC	above
Ponce de Leon to Edgewater	0.34+/-0.09	0.31+/-0.09	0.03+/-0.005	0.03+/-0.01
Edgewater to Oak Hill	0.43+/-0.09	0.35+/-0.14	0.03+/-0.01	0.03+/-0.01
Oak Hill to the Southern Terminus	0.82+/-0.12	0.6+/-0.32	0.02+/-0.01	0.03+/-0.003

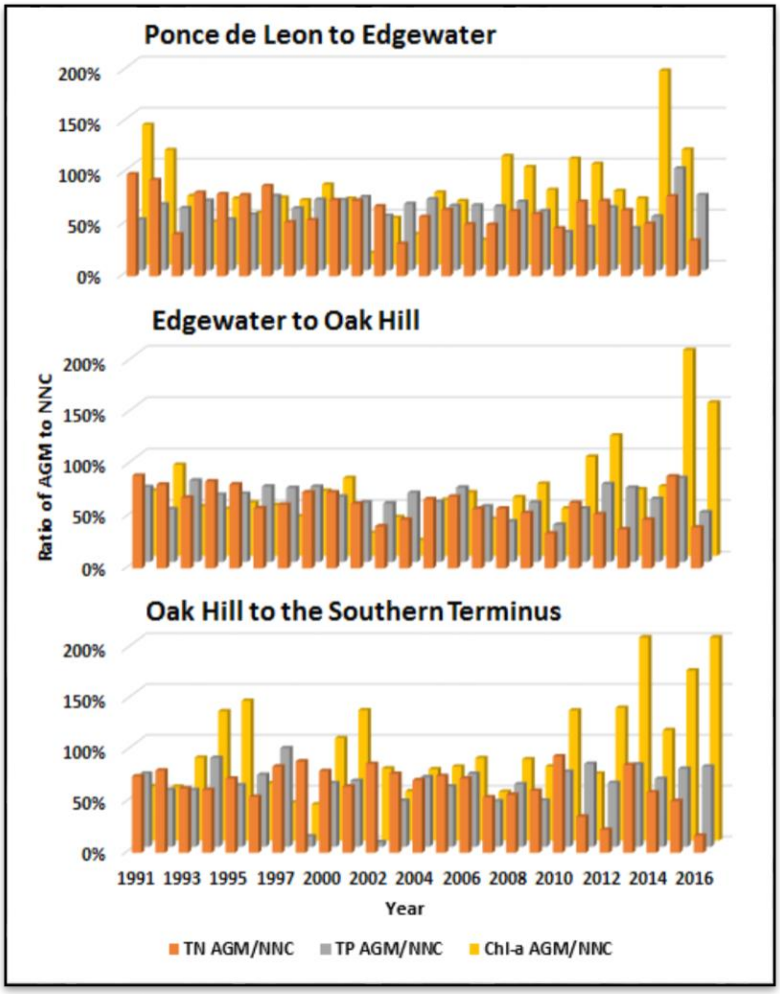

Figure 23. Annual Geometric Means for the Mosquito River Lagoon Segments Showing Elevated Chlorophyll-a, Particularly in Years Following the Initial 2011 Algal Super Bloom.

The NMFS concludes that the TN and TP NNC for the three Mosquito River Lagoon segments (Ponce de Leon to Edgewater, Edgewater to Oak Hill and Oak Hill to the Southern Terminus) are not expected to promote eutrophication because overall, the TN and TP AGM in cases where Chl-a is elevated are within one standard deviation of the TN and TP AGM ifor years where the Chl-a are NNC-compliant (Table 22). Further, based on history of the Mosquito Lagoon basin, instances of elevated Chl-a AGM among TN and TP NNC-compliant AGM are attributable chronic bloom events and climactic factors other than TN and TP that enhance Chl-a levels (see Section 8.1.1).

Among AGM calculated using only TN and TP NNC-compliant sampling events in Pellicer Creek River Estuary, impairment by elevated Chl-a occurred from 2010-2013. The TN and TP AGM in years meeting the TN and TP NNC, but not with elevated Chl-a did not differ substantially from those years with elevated Chl-a (Table 23). These data demonstrate that, when

TN and TP AGM complied with their respective NNC, the Chl-a AGM met the Chl-a NNC in 25 out of 31 years of data (81%, Figure 24).

Table 23. Comparison of Pellicer Creek Average Total Nitrogen and Total Phosphorous Annual Geometric Means where the Chlorophyll-a Geometric Means were Within, Versus Above the Chlorophyll-a Numeric Nutrient Criterion.

	TN mg/L +/-sd	TP mg/L +/-sd
Chl-a within NNC	0.68+/-0.14	0.08+/-0.01
Chl-a above NNC	0.54+/-0.26	0.08+/-0.02

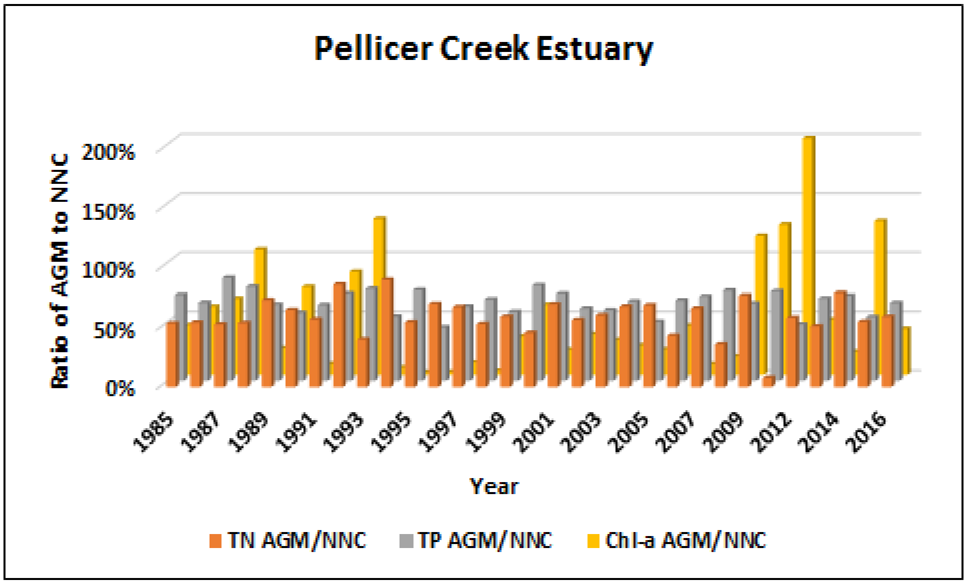

Figure 24. Annual Geometric Means for Pellicer Creek Estuary Showing Elevated Chlorophyll-a with Total Nitrogen and Total Phosphorous Annual Geometric Means within the Numeric Nutrient Criteria Six Times over 31 years.

The NMFS concludes that the TN and TP NNC for the Pellicer Creek Estuary segment are not expected to promote eutrophication because the Chl-a AGM was compliant with the Chl-a NNC in 25 out of 31 years in this segment and the TN and TP AGM for years where the Chl-a AGM was within one standard deviation of TN and TP AGM for years where the Chl-a AGM was within the Chl-a NNC.

The Tomoka River Estuary segment exhibited elevated Chl-a under the TN and TP NNC between 1990 and 1996, with AGMs at 60 to 91 percent of the NNC. Given the 30 years of perfect TN and TP compliance data for this estuary segment (Figure 25), we see that elevated Chl-a is not consistently associated with TN and TP concentrations at or below the NNC. Chl-a AGM was compliant with the Chl-a NNC in 25 out of 30 years in this segment. Figure 25 illustrates that while the TN and TP AGM during the period of elevated Chl-a were relatively close to the NNC, Chl-a was not elevated (e.g., 1984, 1988). The TN and TP AGM in years meeting the TN and TP NNC, but not with elevated Chl-a did not differ substantially from those years with elevated Chl-a (Table 24).

Table 24. Comparison of Tomoka River Estuary Average Total Nitrogen and Total Phosphorous Annual Geometric Means where the Chlorophyll-a Geometric Means were Within, Versus Above the Chlorophyll-a Numeric Nutrient Criterion.

	TN mg/L +/-sd	TP mg/L +/-sd
Chl-a within NNC	0.83+/-0.14	0.09+/-0.02
Chl-a above NNC	0.91+/-0.17	0.09+/-0.01

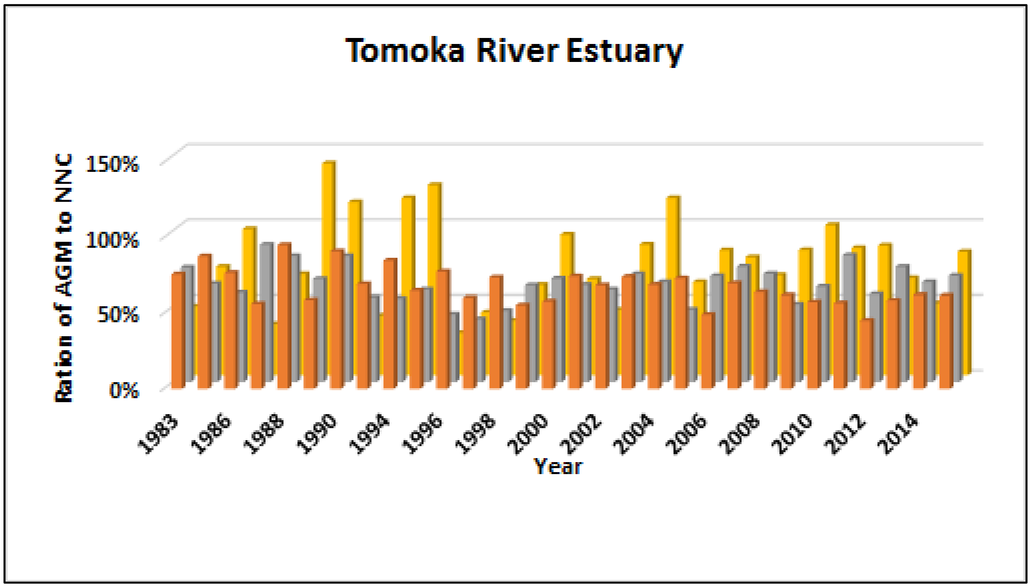

Figure 25. Annual Geometric Means for Tomoka River Estuary Showing Inconsistent Association of Elevated Chlorophyll-a with Total Nitrogen and Total Phosphorous Annual Geometric Means within the Numeric Nutrient Criteria.

The NMFS concludes that the TN and TP NNC for the Tomoka River Estuary segment are not expected to promote eutrophication because the Chl-a AGM was compliant with the Chl-a NNC in 25 out of 30 years in this segment and the TN and TP AGM for years where the Chl-a AGM was elevated was within one standard deviation of TN and TP AGM for years where the Chl-a AGM was within the Chl-a NNC.

Steinhatchee River Estuary had only 5 nonconsecutive years of data with TN and TP AGM meeting their respective NNC: 1991, 1992, and 2014 to 2016. Based on the “as implemented” AGM results, nutrient concentrations appear to have increased in recent years. This segment had elevated Chl-a AGM in 1992, 2015, and 2016 and close to elevated Chl-a AGM in 1991 (94 percent of the Chl-a NNC). In addition, the TN and TP AGM were substantially higher in years with elevated Chl-a (Table 25). In particular, nutrients were higher relative to prior years (60 to 97 percent of the NNC vs 25 to 45 percent of the NNC).

Table 25. Comparison of Steinhatchee River Estuary Average Total Nitrogen and Total Phosphorous Annual Geometric Means where the Chlorophyll-a Geometric Means were Within, Versus Above the Chlorophyll-a Numeric Nutrient Criterion.

	TN mg/L +/-sd	TP mg/L +/-sd
Chl-a within NNC	0.31+/-0.08	0.02+/-0.002
Chl-a above NNC	0.75+/-0.12	0.04+/-0.01

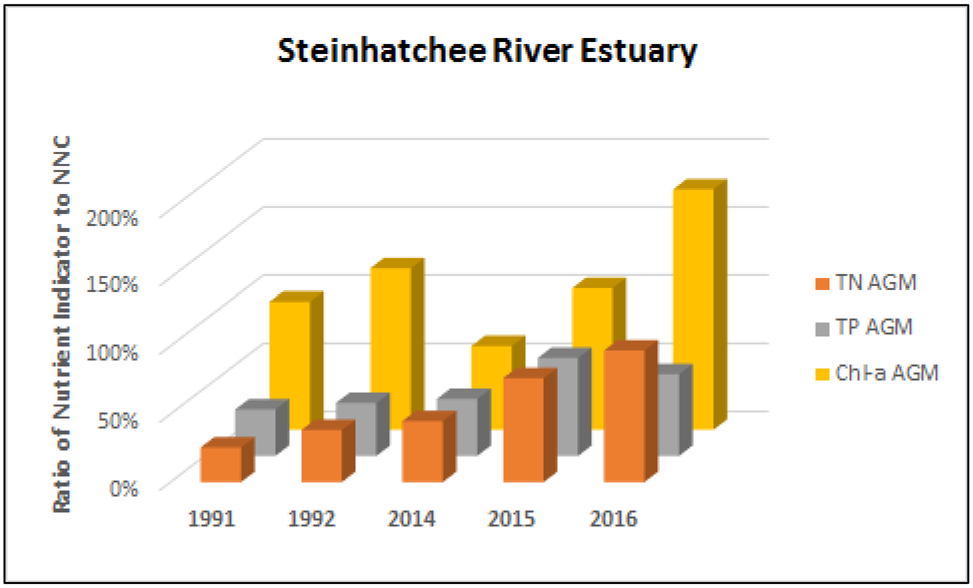

Figure 26. Annual Geometric Means for Steinhatchee River Estuary Showing Association of Elevated Chlorophyll-a with Total Nitrogen and Total Phosphorous Annual Geometric Means within the Numeric Nutrient Criteria.

While three years of TN and TP NNC-compliant AGM were available for the Steinhatchee River Estuary Segment and the plotted AGM suggest a gradual increase in nutrient indicators over the most recent three years (Figure 26), these data may yet be too sparse to arrive at a conclusion on the NNC based on "as implemented" AGM. The segment achieved a Chl-a exceedence when the TN AGM was within 0.02 mg of the NNC, but that AGM includes only two sampling events. The state of Florida requires that the AGM used in determining impairments be representative of year-round conditions. By selecting only those sampling events meeting the TN and TP NNC for the phase two analysis, the Steinhatchee River Estuary dataset was reduced to the point where the underlying variability within the AGM overwhelmed the information contained within the data.

The NMFS concludes that the data available for the Steinhatchee River Estuary is too sparse to effectively evaluate the TN and TP NNC "as implemented."

The PCT-Implemented NNC: Thresholds to be Exceeded in No More than Ten Percent of Observations in a Period of Seven Consecutive Years

For estuary segments with PCT-implemented Chl-a NNC, the NNC is not to be exceeded in more than ten percent of observations over seven consecutive years. Only the Indian River

Lagoon segment between Hobe Sound and St. Lucie had seven or more sequential years of perfect TN and TP NNC-compliant sampling events to allow evaluation of the PCT-based NNC. Evaluation of recent data indicated overall compliance with the Chl-a NNC at the TN and TP NNC. Data collected between 1990 and 1997 indicated Chl-a non-compliance rates above 10 percent, However these data predate the water discharge regulation schedule imposed on the management of Lake Okeechobee, a significant source of nutrient pollution for this system.

> ***The NMFS concludes that, as implemented, the TN and TP NNC for the Indian River Lagoon segment between Hobe Sound and St. Lucie is not expected to promote eutrophication because data collected after regulation of discharges from Lake Okeechobee indicate that the TN and TP AGM within the NNC are consistently associated with Chl-a AGM that are compliant with the Chl-a NNC.***
>
> ***While the absence of the required seven consecutive years of data prevented phase two,"as implemented," evaluation of the PCT-based NNC for St. Marks Offshore and the middle and lower St. Mary's River estuary segments, NMFS concludes that the NNC for these segments are not expected to promote or sustain eutrophication. This conclusion is supported by the phase one analysis showing that the NNC for the middle and lower St. Mary's River estuary segments were not expected to have high frequencies of Chl-a under TN and TP-NNC compliance (Table 12 and Table 13). While data for the St. Marks Offshore segment were too sparse to evaluate the NNC during the phase one analysis, all three observations within the TN and TP NNC were compliant with the Chl-a NNC (Table 13) and most other reference period derived NNC were found to be protective. These observations suggest that the NNC for the St. Marks Offshore estuary segment would also be protective.***

CONCLUSION: DO THE CONCENTRATION-BASED NNC PROMOTE OR SUSTAIN EUTROPHICATION?

With one exception, where sufficient data were available the analyses indicate that the concentration-based NNC developed and proposed by Florida are not expected to promote or sustain eutrophic conditions. This indicates that the methods used in deriving these NNC are sound and we therefore conclude that the NNC derived for those estuary segments that did not have adequate data for our analyses are also not expected to promote or sustain eutrophic conditions.

The exception to this is the Southwest Fork of the Loxahatchee River, which is poorly flushed, receives freshwater discharges that can promote blue green algae growth,and is therefore particularly vulnerable to nutrient effects.

> ***The NMFS concludes that the NNC for the Southwest Fork of the Loxahatchee River potentially promotes or sustains eutrophication based on frequent Chl-a exceedences observed among TN and TP-compliant sampling events for the Southwest Fork of the Loxahatchee River in the phase one analysis and consistent patterns of elevated Chl-a AGM associated with NNC-compliant TN and TP AGM in the phase two analysis.***

Prior to further review in the Response Analysis of this opinion, the presence of ESA listed species under NMFS jurisdiction needs to be confirmed for these segments.

Use of the Southwest Fork of the Loxahatchee River Estuary by ESA-listed species under NMFS' jurisdiction

The Loxahatchee River Estuary of southeast Florida drains a 435 square kilometers watershed and connects to the Atlantic Ocean through the Jupiter Inlet. West of the Jupiter Inlet, the river widens into a large, centralized embayment which opens into three main tributaries: the Northwest Fork, Nork Fork, and Southwest Fork (Figure 27, South Florida Water Management District, 2006). The river's nine segments of the estuary vary in abiotic (e.g., salinity) and biotic (e.g., seagrass presence) characteristics.

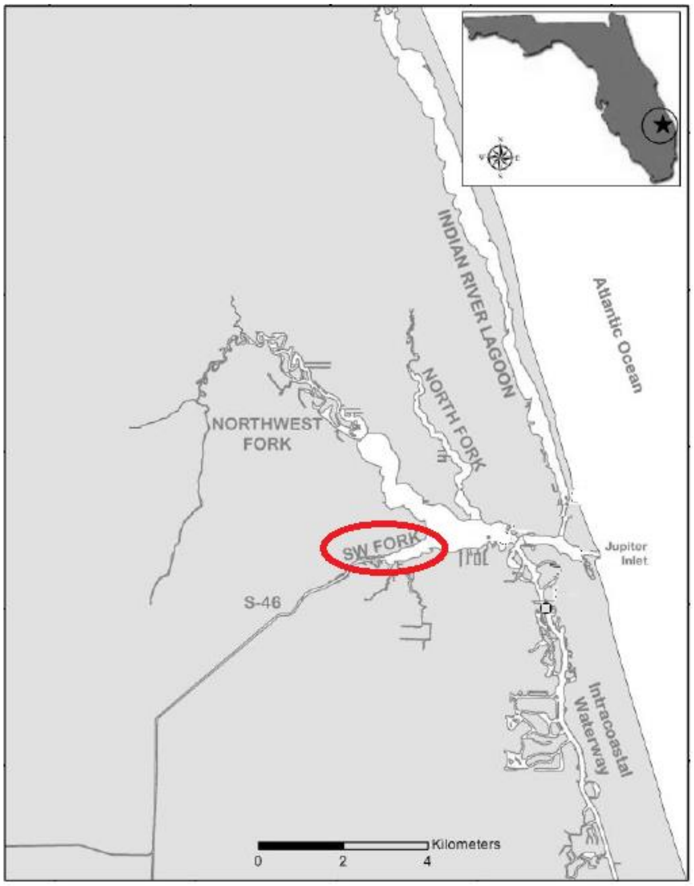

Figure 27. The Loxahatchee River Estuary.

The Southwest Fork of the Loxahatchee River Estuary overlaps with designated critical habitat for loggerhead sea turtle (breeding habitat) and Johnson's seagrass. As indicated in Table 5, the only essential feature of designated critical habitat potentially affected by nutrients is the sargassum element of the designation, and this feature is not likely to occur in the Southwest Fork of the Loxahatchee. Based on the 2011 Draft Loxahatchee River Seagrass Monitoring &

Mapping Report, seagrass beds are patchy-to-absent at the mouth of the tributary (Figure 28), but include continuous areas of Johnson's seagrass (threatened) (Figure 29).

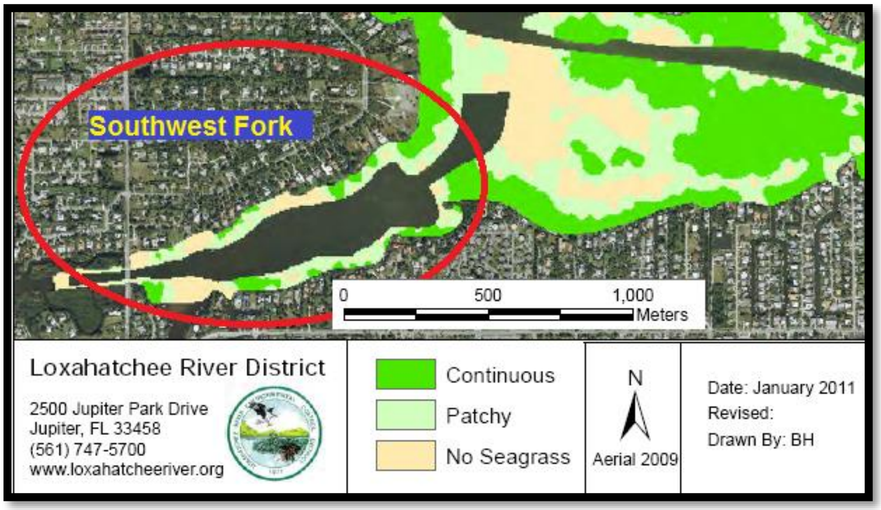

Figure 28. Distribution of Seagrass Beds in the Southwest Fork of the Loxahatchee River Estuary (2010, adapted from Howard et al. (2011)).

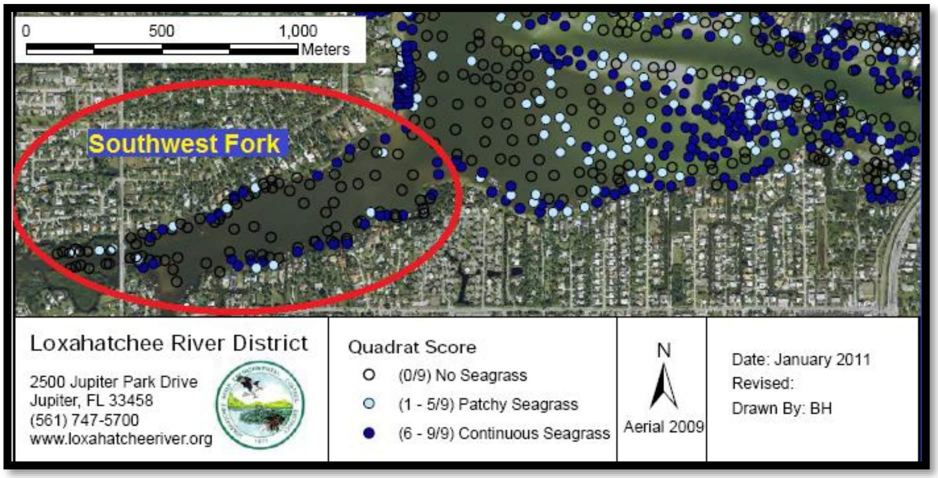

Figure 29. Distribution of Johnson's Seagrass in the Southwest Fork of the Loxahatchee River Estuary (2010, adapted from Howard et al. (2011)).

While seagrass beds are important habitats for green turtles, they are also used by other ESA-listed sea turtle species. However, sea turtles would not be expected to occur frequently in the southwest fork of the Loxahatchee. Only one 2011 stranding of a Hawksbill sea turtle was reported within the segment among monitoring data extending back to 1980 (Florida Fish and Wildlife Conservation Commission - Fish and Wildlife Research Institute Accessed August 3rd 2017). While juvenile grouper use seagrass beds, no Nassau grouper spawning aggregation sites have been reported in Florida. Adult Nassau grouper are rare in Florida waters. Surveys between 2003 and 2013 up the east coast of Florida to Jupiter Inlet report 83 Nassau grouper over 6763 surveys (NMFS 2013a).

Empirical Observations of Water Quality and Seagrass coverage in the Loxahatchee River Estuary: As discussed previously, the Southwest Fork of the Loxahatchee has been characterized by poor water quality for the past decade (Stoner et al. 2016). The Draft Loxahatchee River Seagrass Monitoring and Mapping Report described the 2010 Johnson's seagrass coverage in the estuary as a whole as returning to pre-hurricane disturbance coverage of 20% in 2010 (Howard et al. 2011). According to Howard, as a pioneer species, Johnson's seagrass showed significant increases following hurricanes in 2004, which resulted in scouring or burial of seagrass beds. While the 2015 dataset "Seagrasses in the continental United States as of March 2015" (NOAA 2015) does not identify any seagrass in this segment, this more recent dataset carries the following qualifier: *"This is NOT a complete collection of seagrasses on the seafloor, nor are the locations to be considered exact. The presence and location of the seagrasses have been derived from multiple state websites."* NMFS defers to the direct empirical data reported in (Howard et al. 2011) regarding the presence of this species in the estuary segment.

The only species expected to occur in the Southwest Fork of the Loxahatchee River is Johnson's seagrass. The NMFS concludes that Johnson's seagrass, a threatened species, is likely to be exposed to eutrophication promoted under the NNC proposed for this estuary segment.

8.2.3 Conclusion: Exposure Analysis

Among the concentration-based NNC, there is strong evidence that the standards for the Southwest Fork of the Loxahatchee River Estuary promote eutrophication. Current evidence for the remaining estuaries indicates TN and TP concentrations within the NNC do not promote eutrophication. Where data indicate elevated Chl-a associated with TN and TP at the NNC, other factors, particularly chronic algal bloom events, likely influenced Chl-a AGM (see Section 8.1.1 and discussions for the Indian River and Mosquito River Lagoon segments). Because the majority of evidence supports the NNC as protective against eutrophication, NMFS extends the conclusion that the NNC are protective to those NNC for segments with missing or sparse data: Aucilla Offshore, Aucilla River Estuary, Ochlockonee River Estuary, and Palm Beach County ICWW, Ochlockonee/Alligator Harbor Offshore, Apalachicola Offshore, Econfina River Estuary, and St. Marks Offshore. The methodologies in deriving NNC for these segments do not differ from the methodologies used in deriving the NNC for estuaries found to have sufficiently protective NNC. This extension is strengthened when taking into consideration factors that influence the vulnerability of the Southwest Fork of the Loxahatchee to nutrients. The Southwest Fork of the Loxahatchee River is vulnerable to nutrient impairment due to inputs of freshwater water directed from the north fork of the Loxahatchee. As discussed in Section 8.1.1, freshwater inputs promote phytoplankton growth. Further, the Southwest Fork of the Loxahatchee River s vulnerable to nutrients because it is poorly flushed, allowing nutrients accumulate.

The ranges of green, hawksbill, kemp's ridley, leatherback, and loggerhead sea turtles (Northwest Atlantic DPS) overlap with the Loxahatchee River Estuary. However, considering that only one sea turtle standing has been reported in this estuary segment over 37 years of monitoring, NMFS concludes that the likelihood

of these species being exposed to eutrophic conditions is so low as to be discountable. Therefore, NMFS concludes that EPA's approval of Florida's NNC for the Southwest Fork of the Loxahatchee River is not likely to adversely affect these ESA-listed sea turtle species and these species will not be discussed further in this opinion.

NMFS concludes species that do not occur in the Southwest Fork of the Loxahatchee River will not be exposed to eutrophic conditions under the proposed NNC. Therefore NMFS concludes that that EPA's approval of Florida's NNC will have no effect on North Atlantic right whales, smalltooth sawfish, shortnose sturgeon, Atlantic sturgeon, Gulf sturgeon, Nassau Grouper, and elkhorn, staghorn, rough cactus pillar lobed star mountainous star, and boulder star corals. These species will not be discussed further in this opinion.

The NMFS will evaluate the effects of eutrophication on Johnson's seagrass occurring in the Southwest Fork of the Loxahatchee River Estuary Segment.

8.3 Response Analysis

The response analysis only evaluates effects where the NNC are expected to promote (or sustain) eutrophication. This is limited to the Southwest Fork of the Loxahatchee River Estuary. Accordingly, ESA-listed species under NMFS' jurisdiction, and their associated risk hypotheses, that do not occur in this estuary segment are not considered further in the response analysis. These include the North Atlantic Right Whale, Atlantic and shortnose sturgeon, smalltooth sawfish, Nassau Grouper, and the ESA-listed coral species. Table 26 reviews the risk hypotheses that were retained and those that were eliminated due to the results of the exposure assessment.

The overarching pathways to effects leading to these risk hypotheses were discussed previously in section 8.1.1, *Stressors of the Action.* Excess nutrients accelerate the production and turnover of plant and algal biomass. Accumulation and increased turnover of algal and plant biomass (i.e., death, decay, nutrient release) generates suspended solids in the form of organic particulates and phytoplankton, which directly inhibits the growth of beneficial vegetation such as seagrass by blocking sunlight penetration into the water column (Onuf 1996). Suspended organic matter can eventually accumulate upon and smother plants, animals, and benthic habitat surfaces. Macroalgal blooms stimulated by eutrophic conditions not only reduce sunlight penetration, they can overgrow or displace seagrasses (Valiela et al. 1997), alter stream flow and impair benthic habitat for forage species (Welch et al. 1989, Chessman et al. 1992). Bloom-inflicted mortalities further degrade habitat quality indirectly through altered food webs or hypoxic events caused by the decay of dead animals (Lopez et al. 2008) worsening DO conditions.

Table 26. Risk Hypotheses Eliminated in the Exposure Analysis or Retained for the Response Analysis.

	North Atlantic right whale	Green, hawksbill, Kemp's ridley, leatherback, and loggerhead sea turtles	Atlantic and shortnose sturgeon, smalltooth sawfish, and Nassau grouper	Elkhorn and staghorn corals, boulder lobed and mountainous star corals, pillar coral, and rough cactus coral	Johnson's seagrass
Hypothesis: NNC will support eutrophic conditions that affect the survival and fitness[b] of individuals through:					
lethal and sublethal exposures to ammonia	Species does not occur in affected area	Species are not expected to occur in the Southwest Fork of the Loxahatchee	Species do not occur in the affected area		✓
lethal and sublethal exposures to algal toxins					**No:** Plants are not known to be susceptible to algal toxins
lethal and sublethal DO extremes					✓
lethal and sublethal infections					**No:** Eutrophication is not known to contribute to seagrass disease
lethal and sublethal smothering by algae					✓
altered turbidity/light penetration					✓
Hypothesis: NNC will support eutrophic conditions that have indirect effects to survival and fitness through:					
reduction in extent of habitat area /altered substrate	Species do not occur in the affected area	Species are not expected to occur in the Southwest Fork of the Loxahatchee	Species do not occur in the affected area		✓
reduction in prey species					**No:** autotrophic

Eutrophication also alters the chemical nature of water. Given adequate light, photosynthesis converts carbon dioxide into growth of vegetative biomass: macrophytes, periphyton, and phytoplankton. Photosynthetic consumption of carbon dioxide generates oxygen and increases water pH. The breakdown of plant and algal biomass is mediated by microbes, which consume oxygen during respiration and release carbon dioxide, lowering pH. Plants also respire and consume oxygen, but photosynthesis during the day generates more oxygen than consumed. When photosynthesis pauses at night, plant respiration and microbial decay continues, resulting in a diurnal cycle of peak DO and pH during daylight hours and lowest DO concentrations and pH observed pre-dawn. In addition, direct toxic effects can occur in waters with elevated nitrogen in the form of ammonia, with the more toxic form ammonium more prevalent at high pH.

8.3.1 Effects of Lethal and Sublethal Exposures to Ammonia on Johnson’s Seagrass

Ammonia is a primary product of microbial breakdown of organic nitrogen. Ammonia does not accumulate; it is either oxidized to inorganic species or assimilated by aquatic vegetation. Open estuarine and near shore coastal waters that are not near point sources of ammonia such as sewage outfalls and agricultural operations have typical ammonia concentrations ranging from below detection limits to 0.014 to 0.07 mg total ammonia nitrogen (TAN = NH_4^+ and NH_3) per liter (mg TAN/L), predominantly in the form of ionized ammonia (NH_4^+) (USEPA 2001). Ionized (NH_4^+) and unionized (NH_3) ammonia species exist in water in dynamic equilibrium. It is the unionized form of ammonia, NH_3, which is toxic. Water temperature, pH, and salinity determine nitrogen speciation. EPA's water quality guidelines for the protection of aquatic life incorporate the speciation effects of pH and temperature in the freshwaters ammonia guidelines and the effects of pH, temperature, and salinity in the saltwater ammonia guidelines.

Ammonia Toxicity data for Johnson’s seagrass are not available; however, data for effects on eelgrass serves as a suitable surrogate. Ammonia enrichment was shown to significantly reduce eelgrass growth (Van Katwijk et al. 1997). Experimental eelgrass exposure to ammonium in the water column at concentrations below 25 uM were not found to be toxic. Concentrations as low as 25 uM affected the amount of necrotic tissue (i.e., sublethal effects) and survival (i.e. lethal effects), after 5 weeks of exposure (Van Katwijk et al. 1997). This converts to a sustained concentration of 0.35 mg/L total ammonia nitrogen. Meanwhile ammonia concentrations among 2015-2017 NNC-compliant sampling events are lower than this threshold, averaging 0.13 +/- 0.12 mg/L total ammonia nitrogen.

The NMFS concludes that while the TN and TP NNC proposed for the Southwest Fork of the Loxahatchee River Estuary segment are expected to promote eutrophication and associated ammonia generation, Johnson’s seagrass is not likely to be affected by ammonia because ammonia is not expected to reach concentrations toxic to seagrass.

8.3.2 Effects of Lethal and Sublethal Dissolved Oxygen Extremes on Johnson’s Seagrass

Disruption and declines in ambient DO due to eutrophic conditions affects Johnson’s seagrass indirectly through promoting the generation of toxic sulfides. Burkholder et al. (2007) reviewed the role of eutrophication in seagrass decline. Decoupling of oxygen consumption and production by increasing algal biomass results in a DO regime with extreme fluctuations. Seagrasses, shaded by the canopy of algae become unable to photosynthesize. Cascading effects include smaller leaves, fewer shoots, reduced photosynthetic capacity along with the promotion of sediment sulfide generation. Sulfide is toxic to seagrass, further reducing metabolism, photosynthesis, and growth (Goodman et al. 1995, Stumm and Morgan 1996, Erskine and Koch 2000, Holmer and Bondgaard 2001, Greve et al. 2003). The key to adverse effects of depressed DO in Johnson's seagrass is the severity and duration of depressed DO concentrations and suppression of photosynthesis by reduced light penetration. Low DO levels, in and of themselves, would not have adverse effects on Johnson's seagrass. Continuous DO monitoring data measuring periods where DO is expected to be lowest (i.e., pre-dawn) and highest (i.e., mid-day/peak photosynthesis) provides the best evidence of whether a DO regime in characterized by DO extremes. The data available for the Southwest Fork of the Loxahatchee are taken at a single point in time during the daylight hours. Observation of super saturation in daytime samples (i.e., >100 percent saturation) would provide evidence of a DO regime with extremes. Six percent of DO measurements taken during NNC-compliant sampling events at the Southwest Fork of the Loxahatchee were greater than 100 percent saturated with oxygen (i.e., supersaturated). Oxygen saturation among NNC-compliant sampling events averaged 62.6% +/-24.2. The standard deviation places the upper bound of DO at 87%.

> ***The NMFS concludes that while the TN and TP NNC proposed for the Southwest Fork of the Loxahatchee River Estuary segment are expected to promote eutrophication, Johnson's seagrass is likely to be adversely affected by DO extremes in this estuary segment, because there is some evidence that DO extremes leading to sulfide toxicity occurs under the NNC.***

8.3.3 Effects of Lethal and Sublethal Smothering by Algae on Johnson's Seagrass

Smothering of seagrass occurs when eutrophication progresses to the point where algae accumulate and coat seagrasses, sometimes forming decomposing mats that cover seagrass beds. Water quality within the Southwest Fork of the Loxahatchee is known to be nutrient-impaired with impacts to seagrass beds (Howard et al. 2011, Stoner et al. 2016), but these reports do not indicate that the seagrass habitats present are impaired by algal coverage. Smothering by algae is a consequence of extreme eutrophication, as seen in Tampa Bay in 1970s and 1980s (Figure 30). According to Greening et al. (2014) restoration efforts, including nutrient loading reductions, beginning in the 1980s are achieving pre-disturbance, 1950s conditions.

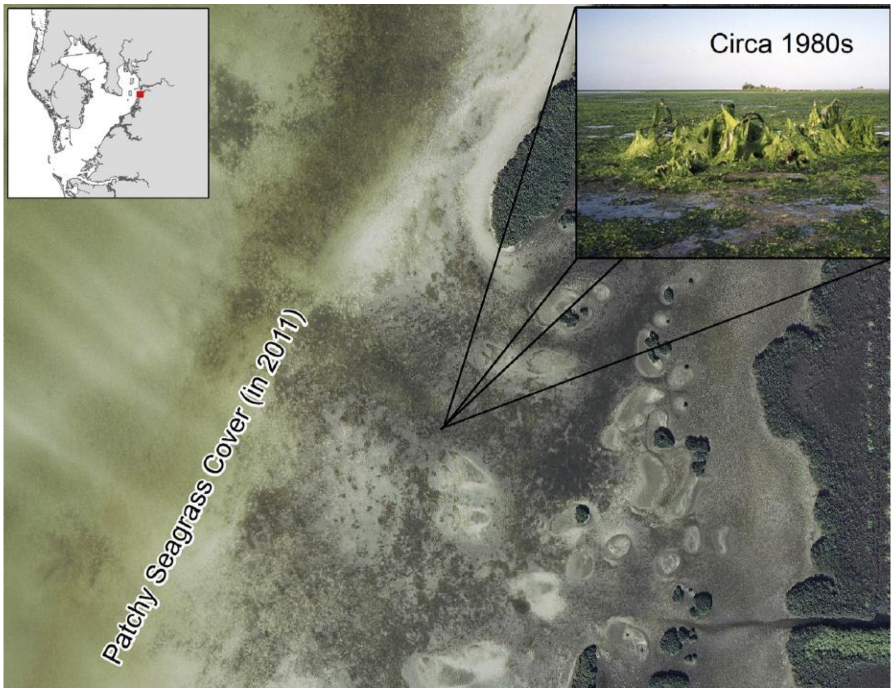

Figure 30. Comparison of 2011 aerial photography showing patchy seagrass beds relative to the same area in Tampa Bay covered by *Ulva* spp. mats circa 1980 (Greening et al. 2014).

Since these conditions do not currently exist in this impaired system, it is reasonable to expect that should conditions in this estuary segment eventually meet the proposed NNC, conditions would, at worst, remain unchanged, and would not lead to smothering.

> ***The NMFS concludes that while the TN and TP NNC proposed for the Southwest Fork of the Loxahatchee River Estuary segment are expected to promote eutrophication, Johnson's seagrass is not likely to be adversely affected by algal smothering because extreme levels of algae such that seagrass beds would be smothered are not expected under the NNC.***

8.3.4 Effects of Altered Light Penetration on Johnson's Seagrass

Eutrophication, as indicated by increased Chl-a, increases water column turbidity and reduces light penetration. The relationship between elevated Chl-a and eutrophication-induced changes in light penetration is well documented. (Gray et al. 2002, Greening and Janicki 2006, Burkholder et al. 2007, Carr et al. 2010, Greening et al. 2011, Howard et al. 2011, Greening et al. 2014, Harding et al. 2014) Some uncertainty is present in this case because current Chl-a data matched with secchi disk measures of light penetration are sparse and a location-specific threshold for acceptable light penetration is not known. Average depths for stations ranged from 0.9 to 1.7 meters. Significant differences between Chl-a compliant observations and observations of elevated Chl-a under the TN and TP NNC were not detected. Light penetration data were sparse, only nine observations among 74 TN and TP compliant sampling events, and the variance among the data was broad, over 35 percent of the mean in stations 72 and 78. Given the well-documented relationship between elevated Chl-a and reduced light penetration, the sparse secchi disk data here are insufficient to support a contrary conclusion that eutrophication associated with elevated Chl-a will not produce the expected reduction of light penetration..

Table 27. Comparison of Light Penetration Data Measured by Secchi Disk.

Station	Chl-a Compliant	Elevated Chl-a
71	1.11+/-0.15 n=6	1.06+/-0.29 n=8
72	1.27+/-0.39 n=5	1.29+/-0.2 n=18
73	0.93+/-0.24 n=4	0.99+/-0.23 n=9
78	1.69+/-0.42 n=4	1.01+/-0.34 n=6
21FLLOX 72	0.9+/-0.1 n=3	1.02+/-0.25 n=23

The NMFS concludes that because the TN and TP NNC proposed for the Southwest Fork of the Loxahatchee River Estuary segment are expected to promote eutrophication, Johnson's seagrass are likely to be adversely affected by reduced light penetration.

8.3.5 Indirect effects: Reduction in extent of Inhabitable area (Altered Substrate) or Reduction in Extent of Useful Habitat for Johnson's Seagrass

The Southwest fork of the Loxahatchee River Estuary Represents 0.52 square kilometers of the 3.9 square kilometer estuary (Figure 31). While the segment itself does not include designated critical habitat for Johnson's seagrass, the estuary drains to the Atlantic Ocean through Jupiter Inlet, which represents a portion of the 91 square kilometer critical habitat designated for Johnson's seagrass (Figure 9). The 2016 Loxahatchee River Water Quality Monitoring Report (Stoner et al. 2016) states that water quality in marine waters of the estuary has been largely good and attributed this to a high degree of flushing through Jupiter Inlet. However, the Chl-a NNC for the Lower Loxahatchee River segment, which includes Jupiter Inlet, is very low, at 1.8 µg/L, and AGM concentrations have slightly exceeded this threshold, but not the NNC thresholds for TN (0.63 mg/L) or TP (0.032 mg/L), in almost every year of the past decade. The NNC for the Southwest Fork of the Loxahatchee River Estuary are considerably higher than those in the lower portion of the estusary: the TN NNC is 1.26 mg/l, the TP NNC is 0.075 mg/L, and the Chl-a NNC is 5.5 µg/L. Since TN and TP levels are not elevated at Jupiter Inlet, with its conservative NNC, we can conclude that the fractional contribution of nutrients from the Southwest Fork of the Loxahatchee is not significantly influencing water quality in this portion of designated critical habitat.

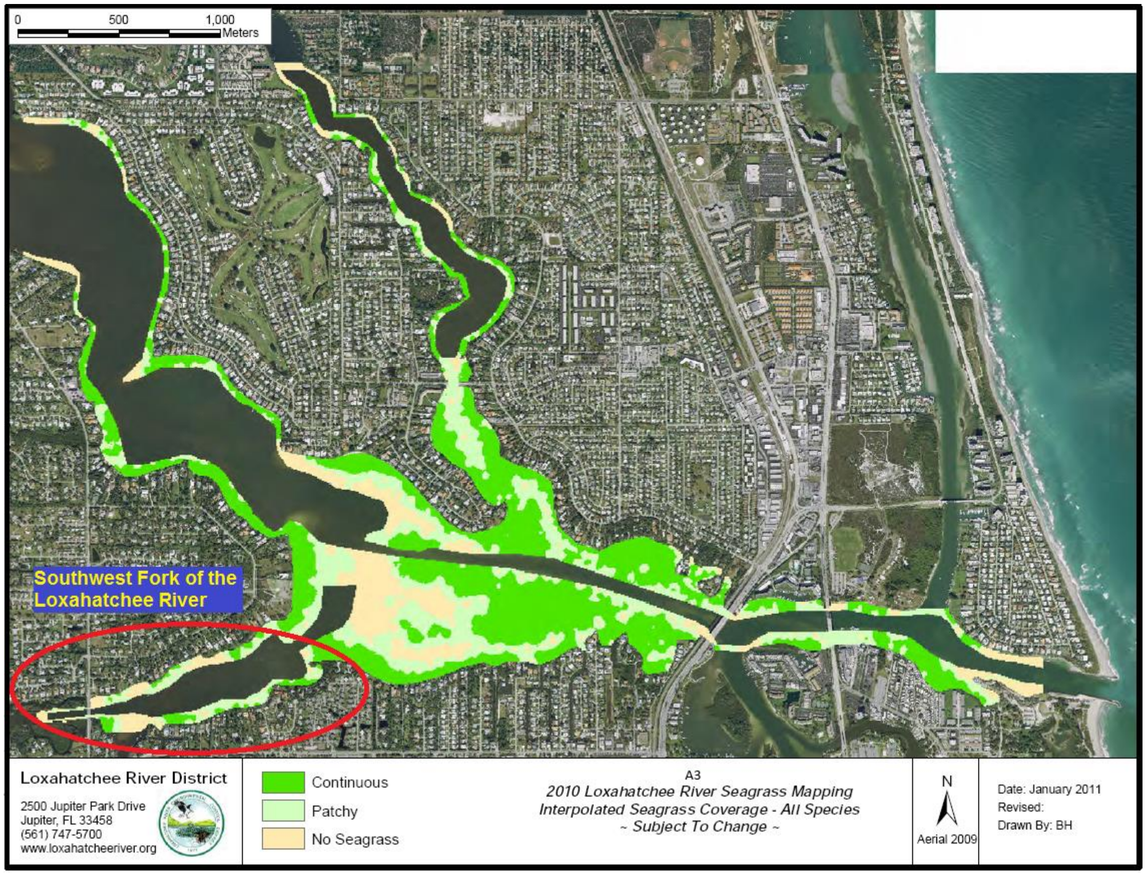

Figure 31. Seagrass Coverage in the Southwest Fork (Circled in Red) of the Loxahatchee River Relative to the Loxahatchee River Estuary.

The NMFS concludes that, while the TN and TP NNC proposed for the Southwest Fork of the Loxahatchee River Estuary segment are expected to promote eutrophication, habitat and designated critical habitat for Johnson's seagrass, a threatened species, are not likely become reduced in extent or habitability because nutrient levels at Jupiter Inlet (designated critical habitat) do not appear to be influenced by conditions in the Southwest Fork of the Loxahatchee (not designated critical habitat) and seagrass coverage in the Southwest Fork accounts for a fraction of the estuary as a whole (Figure 31).

8.4 Risk Characterization Numeric Nutrient Criteria-promoted Eutrophication on Johnson's Sea Grass in the Southwest Fork of the Loxahatchee River Estuary

The risk characterization considers the population-level implications of adverse responses to determine if these are sufficiently large to affect population parameters (e.g., recruitment or reproductive rate). The results of our exposure and response analyses concluded that the NNC-promoted eutrophy in the Southwest Fork of the Loxahatchee River Estuary will have direct adverse effects on Johnson's seagrass through reduced light penetration, reducing photosynthesis and indirect adverse effects through DO extremes leading to generation of toxic sulfides. The response analysis determined that, while eutrophication under the proposed NNC would adversely affect Johnson's seagrass, it would not result in significantly reduced extent or habitability of current occupied habitat or affect nearby designated critical habitat.

The five-year status review for Johnson's seagrass is ten years old (NMFS 2007). However, it indicates that this species had met the first recovery criterion that the species population remain stable for at least 10 years or increase. In 2007, there had been no significant changes in the distribution of this species in the prior 10 years, and, at this time, there are no data suggesting the distribution has changed since 2007. Since publication of the 2007 review, Florida Department of Environmental Protection's Office of Coastal and Aquatic Managed Areas has been monitoring and working with other agencies to recover seagrasses.

> ***NMFS concludes that the proposed TN and TP NNC will adversely affect Johnson's seagrass as they are expected to promote eutrophic conditions in the Southwest Fork of the Loxahatchee, there is evidence that DO extremes and reduced light penetration occur in this waterbody, and existing nutrient impairment conditions in the segment appear to have reduced the extent of seagrass coverage, based on water quality assessments (Stoner et al. 2016) and comparison of surveys from 2007 and 2010 (Howard et al. 2011). However, the NNC are not expected to result in significant effects to the species due to the expected stability of the species overall distribution and size of this estuary segment relative to the extent of seagrass within this estuary (Figure 31) and the range of the species. Further, monitoring data provide evidence that impairment in this segment is not expected to affect nearby designated critical habitat.***

8.5 Uncertainty Analysis for the Numeric Nutrient Criteria Evaluation

The limited availability of data for many estuary segments contributes uncertainty in our assessment of EPA's approval of Florida's proposed NNC. The data available were utilized to the extract the maximum information content to test whether the TP and TN NNC would result in reference Chl-a levels (i.e., the Chl-a NNC), this included data for Loxahatchee indices of eutrophication's effects (i.e., DO saturation and light penetration). The load based NNC are dependent on TMDL implementation. Our evaluation, taken with load reductions reported in annual BMAP status reports, provide evidence that TN and TP loads are reduced under Florida's TMDL program and Chl-a conditions will improve. While there were data gaps among the estuaries with concentration-based NNC, taken together, the data for TN and TP NNC-compliant sampling events in these estuaries contrast markedly with TN and TP NNC-compliant sampling events in the Southwest Fork of the Loxahatchee. The NNC for the Southwest Fork of the Loxahatchee appear to be set at levels that promote eutrophication. NMFS believes that the Chl-a profiles of TN and TP NNC-compliant sampling events in the other segments are more suggestive of background fluctuations. The results of the analyses suggest that, overall, the NNC are not expected to promote eutrophication.

9 CUMULATIVE EFFECTS

"Cumulative effects" are those effects of future state or private activities, not involving Federal activities, that are reasonably certain to occur within the action area of the Federal action subject to consultation (50 CFR 402.02). Future Federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the ESA.

As Florida's population has increased, seagrass habitat has declined from about 5 million acres in the 1950s to 2.2 million acres. The main causes for the decline are coastal engineering (e.g., construction, dredge and fill, beach renourishment), declines in water clarity (e.g., increased turbidity, algal blooms) and substrate disturbance and scarring by vessel traffic (e.g., propeller and prop scars tearing seagrass stems and roots) (FDEP 2017). Commercial and recreational vessel activity in Florida waters is likely to increase in the future with increases in population size, tourism, and average standard of living. As a result, the cumulative effects of vessel damage to bottom substrates, including seagrass beds, is expected to continue to increase. Meanwhile, Florida Department of Environmental Protection's Office of Coastal and Aquatic Managed Areas has been monitoring and working with other agencies to recover seagrasses.

Florida's population has grown steadily throughout the past several decades. From 2000 to 2012 Florida's overall population grew at an annual average rate of 1.5 percent, adding on average 259,600 residents annually (U.S. Census Bureau 2012). Florida is currently the fourth most populous state in the U.S. (~ 20 million residents) and is expected to continue to grow in the decades to come (FDOT 2014). In addition to the large and growing resident population, Florida is the top travel destination that attracts large numbers of tourists and vacationers each year. In 2015 an estimated 101.5 million people visited Florida, an increase of 19 percent since 2011 (Visit Florida official website http:// http://www.visitflorida.org/). General resource demands in Florida are expected to increase as a result of population growth (both resident and visitors), as well as the anticipated increase in the average standard of living in Florida. These demands are particularly high in coastal areas which have higher population densities and greater resource consumption compared to other parts of the state. While the above statements apply to Florida as a whole, the Loxahatchee area itself is relatively rural, making it particularly vulnerable to additional development as Florida grows. The Loxahatchee community had a population of 3,100 people in the 2010 census and, in contrast to Florida's overall growth rate of 1.5 percent, is estimated to have grown by greater than 10 percent to an estimated 3,441 people in 2016 (US Census Bureau 2017).

The future intensity of specific non-Federal activities in the action area is molded by difficult-to-predict future economy, funding levels for restoration activities, and individual investment decisions. However, due to their additive and long-lasting nature, the adverse effects of non-Federal activities that are stimulated by general resource demands, and driven by changes in human population density and standards of living, are likely to compound in the future. Specific human activities that may contribute to declines in the abundance, range, and habitats of ESA-listed species in the action area include the following: urban and suburban development; shipping; infrastructure development; water withdrawals and diversion; recreation, including off-road vehicles and boating; expansion of agricultural and grazing activities, including alteration or clearing of native habitats for domestic animals or crops; and introduction of non-native species which can alter native habitats or out-compete or prey upon native species.

Activities which degrade water quality will continue into the future. These include conversion of natural lands, land use changes from low impact to high impact activities, water withdrawals, effluent discharges, the progression of climate change, the introduction of nonnative invasive species, and the introduction of contaminants and pesticides. Under Section 303(c) of the CWA individual states are required to adopt WQSs to restore and maintain the chemical, physical, and biological integrity of the nation's waters. EPA must approve of state WQSs and this approval is subject to ESA section 7 consultation, which is the purpose of this Opinion. While some of the stressors associated with non-federal activities which degrade water quality will be directly accounted for in section 7 consultations between NMFS and EPA, some may be accounted for only indirectly, while others may not be accounted for at all. In particular, many non-point sources of pollution, which are not subject to CWA NPDES permit and regulatory requirements, have proven difficult for states to monitor and regulate. Non-point source pollution have been linked to loss of aquatic species diversity and abundance, coral reef degradation, fish kills, seagrass bed declines and toxic algal blooms (Gittings et al. 2013). Non-point sources of pollution are expected to increase in Florida as the human population continues to grow. Florida will need to address increases in non-point source pollution in the future to meet the state's approved WQS and designated water body use goals. Given the challenges of monitoring and controlling non-point source pollution and accounting for all the potential stressors and effects on listed species, chronic stormwater discharges will continue to result in aggregate impacts.

9.1 Planned Restoration Efforts in the Loxahatchee Estuary

Restoration efforts in the Loxahatchee seek to increase base flows into the Northwest Fork while not compromising the ecological integrity of downstream reaches (i.e., estuary) nor impairing valued ecosystem components of the estuary such as oysters and seagrasses. A "Reasonable Assurance Plan" or RAP is under development for the Loxahatchee River Estuary. As explained by Florida on its website (http://www.dep.state.fl.us/water/watersheds/rap.htm):

> *EPA regulations allow states to place certain impaired water bodies into Category 4b of the Integrated Report Categories instead of category 5 – Impaired and needs a TMDL. The Florida Watershed Restoration Act (Section 403.067(4) F. S.) explicitly allows DEP to not list impaired waters under Category 5 if they already have control programs in place that will assure that water quality standards will be restored. These types of waterbodies – impaired, but with control programs already being implemented to reduce pollutant loadings – are placed in assessment Category 4b for Clean Water Act section 303(d) reporting purposes.*

The Loxahatchee River Watershed Restoration Project, implemented by the U.S. Army Corps of Engineers and The South Florida Water Management District, includes a monitoring plan and adaptive management plan to ensure the monitoring data inform any adjustments that need to be made in the implementation of the restoration methods selected (Foster 2017). The stated project objectives are:

- Objective 1: Restore wet & dry season flows of water to the Northwest Fork of the Loxahatchee River and the river floodplain
- Objective 2: Restore estuarine communities in the Loxahatchee River Estuary
- Objective 3: Increase spatial extent & function of remaining natural areas
- Objective 4: Restore connections among the Loxahatchee River headwater natural areas to improve hydrology, sheetflow, hydroperiods, natural storage, & vegetation communities
- Objective 5: Restore native plant & animal species abundance & diversity

The effort would correct conditions that contribute to poor water quality within the entire estuary, including the Southwest Fork of the Loxahatchee River. This selection of restoration methods is underway.

10 INTEGRATION AND SYNTHESIS

The *Integration and Synthesis* section is the final step in our assessment of the risk posed to species and designated critical habitat as a result of implementing the proposed action. In this section, we add the effects of the action from the *Risk Characterization* section of this opinion to the *Environmental Baseline* and the *Cumulative Effects* to formulate the agency's opinion as to whether the proposed action is likely to: (1) reduce appreciably the likelihood of both the survival and recovery of a ESA-listed species in the wild by reducing its numbers, reproduction, or distribution; or (2) reduce the value of designated designated critical habitat for the conservation of the species. These assessments are made in full consideration of the status of the species and designated critical habitat (Section 5).

The case for excess nutrients and Chl-a in the Southwest Fork of the Loxahatchee has recently been described in detail in the Loxahatchee River Water Quality Monitoring Report (Stoner et al. 2016). Survey maps in the Draft Loxahatchee River Seagrass Monitoring & Mapping Report (Howard et al. 2011) suggests an overall decline in seagrass coverage in this estuary segment, but not the estuary as a whole. The analysis in this opinion confirms that the TN and TP NNC result in elevated Chl-a and, although inadequately monitored, other data suggest eutrophic conditions occur that would directly adversely affect Johnson's seagrass: DO extremes and reduced light penetration. Poor flushing of the segment leading to increased mean water residence times makes the system more vulnerable to eutrophication because this favors accumulation of nutrients and subsequent stimulation of photosynthesis/Chl-a. Further, the flood control structure directs freshwater into the segment, which enhances growth of blue green algae. To remedy this, the NNC would need to be revised to address this vulnerability or the flushing regime of the system would need to be altered such that the segment is no longer vulnerable. While periods of reduced salinity promote algal growth, we also must consider that reduced salinity is directly lethal to Johnson's seagrass and has been identified as a serious threat to the species (Torquemada et al. 2005a, Kahn and Durako 2008, Griffin and Durako 2012). Using mesocosm experiments, Gavin and Durako (2014) determined that prolonged pulses of reduced salinity (two weeks at 10 ppt followed by one week at 25 ppt) reduced survival greater than 50 percent among Johnson's seagrass collected near the mouth of the Southwest Fork of the Loxahatchee. Stormwater events in Florida, and within the Loxahatchee River Estuary itself, have resulted in large fluctuations in salinity and occasional sustained periods of salinity <15 ppt (Russell and McPherson 1983, Gavin and Durako 2014).

The question before us is whether the effects of the DO extremes and reduced light penetration will reduce appreciably the likelihood of both the survival and recovery of Johnson's seagrass by decreasing its numbers, reproduction, or distribution. As described in section 6.5 and in Virnstein et al. (2009), Johnson's seagrass is a pioneer species characterized by highly dynamic distribution that is patchy in both space and time, expanding and contracting according to availability niche spaces among other species, with some avoidance of taller canopy forming species. The species' occurrence in the estuary in the 2007 surveys was attributed to a post hurricane colonization pulse by this pioneer species, and the subsequent decline was interpreted as a return to pre-hurricane levels in 2010 (Howard et al. 2011). The segment represents only a small portion of Johnson's seagrass overall range and coverage within the Loxahatchee Estuary (Contrast Figure 9, Figure 27, and Figure 31). The segment also currently provides poor habitat for seagrass species. Restoration efforts are planned that would improve the habitat quality of the segment for seagrass in the estuary, but these actions are not certain to occur (Foster 2017). Restoration would be expected to correct the hydrological regime in the Southwest Fork, thereby attenuating stormwater-induced extreme fluctuations (i.e., declines) in salinity that negatively affect seagrass survival and persistence and promote growth of blue-green algae. A corrected hydrological regime would also increase water turnover rates in the system, reducing nutrient retention.

The NMFS took the two environmental monitoring reports (Howard et al. 2011, Stoner et al. 2016), the Watershed Restoration Project Plan (Foster 2017), the five year status review (NMFS 2007) and the results of our analysis into consideration in making its final determination.

> ***The NMFS concludes that, while the TN and TP NNC for the Southwest Fork of the Loxahatchee River are expected to promote eutrophic conditions, they are not expected to jeopardize the continued existence of Johnson's seagrass or adversely modify critical habitat designated for this species. The NMFS arrives at this conclusion based on the limited importance of this estuary segment relative to the overall distribution of the species, the marginal quality of the existing habitat within the segment due to freshwater inputs, and the expectation that the distribution of the species is currently stable within this estuary and within its range (NMFS 2007, Howard et al. 2011).***

NMFS also concludes that the NNC will not adversely modify critical habitat designated for Johnson's seagrass because monitoring data provide evidence that conditions in the segment do not significantly affect the nearby designated critical habitat in Jupiter Inlet.

11 CONCLUSION

After reviewing the current status of the ESA-listed species, the environmental baseline within the action area, the effects of the proposed action, and cumulative effects, it is NMFS' opinion that the proposed action is not likely to jeopardize the continued existence or recovery of North Atlantic right whale, green, hawksbill, Kemp's ridley, Leatherback, or loggerhead sea turtle, smalltooth sawfish, shortnose or Atlantic sturgeon, Atlantic sturgeon, Nassau grouper, elkhorn, staghorn, rough cactus, pillar, lobed star, mountainous star, or boulder star coral, or Johnson's seagrass or to destroy or adversely modify designated critical habitat for the North Atlantic right whale, smalltooth sawfish, loggerhead sea turtle, elkhorn or staghorn coral, or Johnson's seagrass. NMFS anticipates Johnson's seagrass (threatened) is likely to be adversely affected by DO extremes and reduced light penetration resulting from eutrophic conditions under the NNC.

12 INCIDENTAL TAKE STATEMENT

Section 9 of the ESA and Federal regulations pursuant to section 4(d) of the ESA prohibit the take of endangered and threatened species, respectively, without a special exemption. Section 7(b)(4) and section 7(o)(2) of the ESA provide that taking that is incidental to an otherwise lawful agency action is not considered to be prohibited taking under the ESA if that action is performed in compliance with the terms and conditions of an ITS. The only species found to be adversely affected in this consultation is Johnson's seagrass (threatened) and sections 7(b)(4) and 7(o)(2) of the Act generally do not apply to listed plant species. However, limited protection of listed plants from take is provided to the extent that the ESA prohibits the removal and reduction to possession of Federally listed endangered plants or the malicious damage of such plants on areas under Federal jurisdiction, or the destruction of endangered plants on non-Federal areas in violation of State law or regulation or in the course of any violation of a State criminal trespass law. The applicable law in this case is The Florida Endangered and Threatened Species Act (§379.229). In addition, Florida's Wildlife Rule, Chapter 68 in the Florida Administrative Code 68A-27.003, states that no person shall pursue, molest, harm, harass, capture, possess, or sell any endangered species or parts thereof or their nests or eggs except as authorized by specific permit.

13 CONSERVATION RECOMMENDATIONS

Section 7(a)(1) of the ESA directs Federal agencies to use their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of the threatened and endangered species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a proposed action on ESA-listed species or designated critical habitat, to help implement recovery plans or develop information (50 CFR 402.02).

The following conservation recommendations would provide information for the refinement of water quality criteria for the protection of ESA-listed species under NMFS' jurisdiction:.

- Coordinate with NMFS, when requested, to support NMFS's review and communication with Florida about the State's conclusions regarding possible future 303(d) listings or delistings for chlorophyll a in segments where threatened and endangered species and/or their designated critical habitat occur.
- Coordinate with NMFS, as requested, until all basins in Florida have been assessed for NNC under the 5 year Strategic Plan or until a mutually agreed upon termination by EPA and NMFS, to support NMFS's review and evaluation of monitoring data from waters where ESA-listed species and designated critical habitat under NMFS' jurisdiction occur. These data will include, at a minimum, date, time, monitoring station identification, monitoring station coordinates in decimal degrees, Chl-a, TP, TN (or constituent nitrogen species), and DO. The monitoring data will be used to proactively identify emerging or masked issues and evidence of eutrophy (i.e., elevated Chl-a) under TN and TP levels that are compliant with NNC.
- In the event that an analysis of monitoring data indicates eutrophic conditions under the NNC, coordinate with NMFS to communicate these concerns and make recommendations to the state regarding the criteria and how to bring the waterbody back into compliance with the Chl-a NNC.
- Conduct or support research expanding scientific understanding of the linkages between eutrophication in marine ecosystems and disease of aquatic biota.
- While NMFS recognizes that EPA has no direct involvement or responsibility for the RAP, EPA could condition its approval of the NNC for the Southwest Fork of the Loxahatchee River upon the successful RAP development and implementation.

In order for NMFS' Office of Protected Resources ESA Interagency Cooperation Division to be kept informed of actions minimizing or avoiding adverse effects on, or benefiting, ESA-listed species or their designated critical habitat, EPA should notify the ESA Interagency Cooperation Division of any conservation recommendations they implement in their final action.

14 REINITIATION OF CONSULTATION

This concludes formal consultation for EPA's approval of Florida's NNC, DO Criteria, and Turbidity Limits. As 50 CFR 402.16 states, reinitiation of formal consultation is required where discretionary Federal agency involvement or control over the action has been retained (or is authorized by law) and if: (1) the amount or extent of incidental take is exceeded, (2) new information reveals effects of the agency action that may affect ESA-listed species or designated critical habitat in a manner or to an extent not considered in this opinion, (3) the agency action is subsequently modified in a manner that causes an effect to the ESA-listed species or designated critical habitat that was not considered in this opinion, or (4) a new species is ESA-listed or designated critical habitat designated that may be affected by the action.

Existing data for other estuary segments did not indicate elevated Chl-a under their respective TN and TP NNC. With the exception of the Southwest Fork of the Loxahatchee River, our overarching conclusion on Florida's NNC are not expected to promote or sustain eutrophic conditions where effects to ESA-listed species are under NMFS's jurisdiction, and therefore are not anticipated to adversely affect the other ESA-listed species considered in this biological opinion. The NNC for these waters were not evaluated further in this opinion because the data in hand at the time of the analysis did not indicate eutrophic conditions would be promoted or supported where ESA-listed species under NMFS' jurisdiction occur. However, the current monitoring data are more limited, and in some instances absent, for some estuaries. Additional data for these waters will be generated as Florida's strategic monitoring plan is carried out. Persistence of existing eutrophic conditions or an increase in the extent or frequency of Chl-a excursions over Chl-a NNC under nutrient conditions that are compliant with the TN and TP NNC would indicate the existing TN and TP NNC for these segments support eutrophy, and associated hypoxia, that is expected to result in take due to the adverse effects of hypoxia as described in this opinion. This would constitute "new information" and would trigger reinitiation.

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